abstracted by LiSCos
Del. 3-6-70. jal

14010EMS09/69

DAST - 16

PB189758

W70-04006

BULK TRANSPORT OF WASTE SLURRIES TO INLAND AND OCEAN DISPOSAL SITES

VOLUME II

CRITERIA FOR WASTE MANAGEMENT

COLOR ILLUSTRATIONS REPRODUCED
IN BLACK AND WHITE

BECHTEL

September 1969

Reproduced by the
CLEARINGHOUSE
for Federal Scientific & Technical
Information Springfield Va. 22151

194

LIST OF TABLES

3.1.1 Great Lakes Study Region - Projected Populations.

3.1.2 Atlantic Coast Study Region - Projected Populations.

3.1.3 Projected Digested Sludge Quantities

3.2.1 Great Lakes Study Area - Maintenance Dredging Quantities Summary Sheet.

3.2.2 Great Lakes Study Area - Maintenance Dredging Quantities.

3.2.3 Atlantic Coast Study Area - Maintenance Dredging Quantities.

3.3.1 Great Lakes Study Area - Current Power Plant Coal Burn and Estimated Fly Ash Production.

3.3.2 Atlantic Coast Study Area - Current Power Plant Coal Burn and Estimated Fly Ash Production.

3.3.3 1966 Production and Utilization of Ash.

3.3.4 Fly Ash Utilization in the United States.

3.4.1 Average Turbidities and Estimated Solids Production of Selected U. S. Water Supplies Employing Coagulation.

3.4.2 Projected Waste Quantities - Chemical Coagulation Residues.

4.1.1 Unit Sludge Process Costs.

4.1.2 Total Sludge Disposal Costs.

4.3.1 Estimated 1967 Fly Ash Land Disposal Costs.

5.2.1 Typical Element Concentrations and Properties for Digested Sludge.

LIST OF FIGURES

3.1.1 Daily Residue Production from a Typical 1 MGD Municipal Waste-Water Treatment Plant.

3.1.2 Great Lakes Megalopolis.

3.1.3 Atlantic Coast Megalopolis.

4.1.1 Cost of Sludge Disposal by Various Methods for City of 10,000 people.

4.1.2 Cost of Sludge Disposal by Various Methods for City of 100,000 people.

4.1.3 Cost of Sludge Disposal by Various Methods for City of 1,000,000 people.

4.2.1 Operational Sketch of Hopper Dredge.

6.2.1 Surf Clam Survey Cruises.

7.2.1 Heat Treatment of Digested Sludge.

8.3.1 Proposed Distribution System for Sludge Waste Management

ACKNOWLEDGEMENTS

We wish to acknowledge the contributions of the following people in compiling this report:

P. E. Snoek of Bechtel Corporation for the writing and editing of the report, and Messrs. W. J. Stump and C. L. Amos of Bechtel Corporation for their assistance in the report writing; Dr. R. I. Dick, Dr. T. D. Hinesly, and Dr. J. T. O'Connor of the University of Illinois for their contributions on land application aspects; Dr. J. Carpenter of Chesapeake Bay Institute, The Johns Hopkins University for his contribution on the environmental aspects of ocean disposal; Dr. Iraj Zandi, Towne School of Engineering University of Pennsylvania for his contributions on treatment and transportation of bulk wastes; Mr. John J. Wirts, Superintendent, Easterly Pollution Control Centre, Cleveland, Ohio, for information on present methods and costs of disposal.

We also wish to gratefully acknowledge the time and effort contributed by representatives of the following organizations:

American Water Works Association
California Institute of Technology
Chicago Metropolitan Sanitary District
Los Angeles Hyperion Sewage Treatment Plant
National Ash Association
New York City Environmental Protection Association
U. S. Army Corps of Engineers
Woods Hole Oceanographic Institution

E. J. Wasp
Project Manager

VOLUME II

CRITERIA FOR REGIONAL WASTE MANAGEMENT

PREFACE

1.0 INTRODUCTION

2.0 SUMMARY AND CONCLUSIONS

3.0 WASTE QUANTITIES

  3.1 DIGESTED SLUDGE

    3.1.1 Suspended Solids Contribution
    3.1.2 Fresh Sludge Production
    3.1.3 Tertiary Treatment
    3.1.4 Digestion Losses
    3.1.5 Standard Metropolitan Statistical Areas
    3.1.6 Population Projections
    3.1.7 Sewered Populations
    3.1.8 Projected Sludge Quantities

  3.2 MAINTENANCE DREDGINGS

    3.2.1 Historical Record
    3.2.2 Projected Waste Quantities

  3.3 FLY ASH

    3.3.1 Projected Coal Burn
    3.3.2 Utilization
    3.3.3 Projected Waste Quantities

  3.4 WATER TREATMENT PLANT RESIDUES

    3.4.1 Solids Production
    3.4.2 Surface Water Consumption
    3.4.3 Projected Chemical Coagulation Wastes

4.0 PRESENT METHODS AND COSTS OF DISPOSAL

  4.1 DIGESTED SLUDGE

    4.1.1 Disposal to the Land
    4.1.2 Disposal to Water
    4.1.3 Disposal to the Air
    4.1.4 Disposal Costs

- 4.2 MAINTENANCE DREDGINGS
  - 4.2.1 Removal to Deep Water
  - 4.2.2 Disposal to Landfills and Offshore Dike Areas
  - 4.2.3 Disposal Costs
- 4.3 FLY ASH
  - 4.3.1 Disposal Methods
  - 4.3.2 Disposal Costs
- 4.4 WATER TREATMENT PLANT RESIDUES
  - 4.4.1 Chemical Coagulation Wastes
  - 4.4.2 Filter Backwash Water

5.0 ENVIRONMENTAL CONSIDERATIONS FOR REGIONAL LAND DISPOSAL SYSTEMS

- 5.1 EFFECT ON WATER ENVIRONMENT
  - 5.1.1 Pathogenic Microorganisms
  - 5.1.2 Viruses
  - 5.1.3 Chemical Additions
- 5.2 EFFECT ON SOILS AND CROPS
  - 5.2.1 Chemical Additions
  - 5.2.2 Historical Record
- 5.3 OTHER ENVIRONMENTAL EFFECTS
  - 5.3.1 Airborne Pollution
  - 5.3.2 Insect Vectors
  - 5.3.3 Crops as Vectors
  - 5.3.4 Soil Contact
  - 5.3.5 Nuisance Conditions

6.0 ENVIRONMENTAL CONSIDERATIONS FOR REGIONAL OCEAN DISPOSAL SYSTEMS

- 6.1 FACTORS OF CONCERN
- 6.2 DILUTION REQUIREMENTS
  - 6.2.1 Transparency
  - 6.2.2 Nutrients
  - 6.2.3 Dissolved Oxygen Concentration

6.3 SHELLFISH CONTAMINATION
    6.3.1 Viruses
    6.3.2 Heavy Metals
6.4 CONCLUSIONS

7.0 TREATMENT OF WASTE MATERIALS
7.1 EVALUATION OF NEED
7.2 TREATMENT FOR COMMUNICABLE DISEASE AGENTS
    7.2.1 Storage
    7.2.2 Heat Treatment
    7.2.3 Chemical Disinfection
    7.2.4 Radiation and Other Physical Methods
7.3 TREATMENT FOR NITROGEN REMOVAL
    7.3.1 Nitrogen Removal Experience
    7.3.2 Removal Prior to Land Application
    7.3.3 Removal in Situ
    7.3.4 Removal from Leachate

8.0 LAND DISTRIBUTION SYSTEMS
8.1 STATE OF THE ART
8.2 APPLICATION RATES
8.3 METHODS OF APPLICATION
    8.3.1 Sludge
    8.3.2 Dredgings
    8.3.3 Fly Ash
8.4 CONCLUSIONS

9.0 OCEAN DISTRIBUTION SYSTEMS
9.1 STATE OF THE ART
9.2 ATLANTIC COAST ALTERNATIVES
    9.2.1 Barge and Tanker Disposal
    9.2.2 Pipe and Diffuser Disposal
9.3 CONCLUSIONS

APPENDIX A

BIBLIOGRAPHY

PREFACE

Contract No. 14-12-156 was issued by the Federal Water Pollution Control Administration of the Department of the Interior to Bechtel Corporation on April 10, 1968. The purpose of the contract was to determine the technical feasibility and economics of transporting sewage treatment plant sludge and dredging spoils by currently available transport systems for both land and ocean disposal. To insure the general application of this concept, regional considerations were emphasized. Two additional bulk waste materials, fly ash and water treatment plant sludge, were also investigated for possible inclusion in the disposal system.

The report summarizing the work under this contract has been presented in three volumes. This volume is concerned mainly with environmental criteria for regional waste management systems. Volume I presents a detailed evaluation of two typical regional systems, the Northeast Ohio land disposal system and the Baltimore-Washington ocean disposal system. Included is a recommended demonstration plan of the recommended bulk transport and disposal concept. Volume III is devoted to pipelining of waste materials, with major reference to a test program conducted under this study.

1.0 INTRODUCTION

This volume seeks to establish basic environmental criteria for disposal of selected waste materials, including sewage treatment plant sludge, maintenance dredgings, power plant fly ash, and filter plant residues. It examines both land and ocean disposal alternatives in terms of a general solution for two broad regions, namely the Great Lakes Megalopolis from Milwaukee to Buffalo and the Atlantic Coast Megalopolis from Boston to Norfolk. Sludge disposal is stressed because this is a widespread major problem while the disposal of other wastes is acute only in some areas. Furthermore, sludge disposal presents most of the major problems that will occur with disposal of any of the four wastes considered. Waste quantities are projected to the year 2000, but the study conclusions are judged to be applicable for a much longer time span. Present methods and costs of disposal on all four wastes have been included to complete the picture of the overall waste disposal problem.

2.0 SUMMARY AND CONCLUSIONS

2.1 THE PROBLEM

Disposal of waste materials today is a major national problem. This is the result of the ever increasing growth of population and concurrent expansion of industrial output. In particular, disposal of sludge from large municipal sewage treatment plants has become very troublesome and expensive. Furthermore, the severity of this problem can be expected to increase in the future. As shown in the table below, waste quantities are predicted to generally increase from now to the year 2000. However, digested sludge loads will nearly triple, due to population growth, wider use of garbage grinding, and higher levels of wastewater treatment.

Projected Waste Quantities
dry tons/day

	1968	1975	2000
Great Lakes Region			
Digested Sludge	1350	1670	3760 (1)
Maintenance Dredging	16,700	16,700	16,700
Power Plant Fly Ash	9470	11,480	5920
Filter Plant Residue	217	247	407
Atlantic Coast Region			
Digested Sludge	2190	2740	6120 (1)
Maintenance Dredging	51,000	54,800	67,500
Power Plant Fly Ash	7320	8520	3040
Filter Plant Residue	173	197	323

(1) Includes waste chemical sludge from tertiary treatment

2.2 INADEQUACY OF PRESENT METHODS OF DISPOSAL

Concurrent with rising waste volumes is a steadily decreasing number of suitable disposal alternatives. An example is sludge disposal for large cities. Increased urbanization of the area surrounding the treatment plant has nearly eliminated lagooning as an attractive method of disposal.

Municipal and industrial incineration of solid wastes has been widely practiced for many years. Sludge incineration has gained wide acceptance only in the last few decades. However, the increasing awareness of society as to the effect of all forms of pollution on man's environment leads one to doubt whether this can be considered the ultimate solution. Conversion of pollutants from the solid or liquid state to the gaseous form is obviously not an answer. Although present technology can result in minimal emission of particulate matter and odors, higher air pollution standards will require control of other emissions which are not yet classed as pollutants. Even now, the oxides of nitrogen are being singled out as the next contaminant for further controls as demonstrated by California's proposed "pure air act". Because their role is fundamental to the formation of photochemical smog, as well as contributing to plant damage, federal air quality guidelines for limiting oxides of nitrogen from stationary sources may be expected in the near future. At present, neither removal from stack gases nor reduction through modification of the combustion process appears feasible without incurring a significant economic penalty.

Furthermore, tertiary treatment of municipal sewage may result in a substantial increase in the volume of inorganic material for disposal. The resulting change in sludge properties will add to the fuel cost while increasing the amount of ash for disposal. Thus, sludge incineration may become increasingly unattractive. Other combustion processes are in the same situation. All face increasing restriction on air pollution and a waste that is becoming less suited to this method of disposal. In some cases additional problems may develop like the formation of nutrient rich liquors, which if recycled back through the plant, can result in lowering the quality of the effluent.

A similar situation exists with disposal of other wastes. Maintenance dredgings have traditionally been discharged to nearby open water as a matter of economy. Currently there are strong objections to continuing this policy, particularly in the Great Lakes region. Although the effects of dredgings disposal on the receiving area are still largely unknown, the material is sometimes heavily polluted and so this procedure must be considered, at least in these cases, as being conditionally unacceptable.

In the case of fly ash from power plants, land fill in nearby areas is currently the most widely used method of disposal. This procedure is acceptable from an environmental standpoint and is economic as well. However, the increase in land use over the next few decades will reduce the feasibility of using this alternative in many metropolitan areas. In spite of increased utilization, it is anticipated that waste quantities will increase until 1985. After that year, nuclear power growth should result in a falloff in power plant coal burn.

Chemical coagulation residues from rapid sand filter plants are usually discharged back to the water source. In some cases, the residue is discharged to sanitary sewers where it eventually ends up as part of the waste-water treatment plant sludge. The first method is destined to decrease in use over the near future, as obviously, this results in some degradation of the receiving water. The second alternative is just a transfer of the problem from one place to another.

2.3 DISPOSAL SYSTEM REQUIREMENTS

Any waste disposal system of major consequence to the total national effort in the field of pollution control should satisfy the following:

1. Prevention of any form of environmental degradation.
2. Plan shall be applicable to a large section of the country.
3. Plan shall be capable of handling the expected increase in wastes over the near future, with flexibility for further expansion as required.

2.4 LAND AND OCEAN DISPOSAL

Ultimate disposal of waste materials to the land or the ocean shelf has often been advocated. In the case of sewage treatment plant sludge, stabilization is a necessary requirement. The process results in the breakdown of the degradable organic matter and formation of relatively inert organic and inorganic compounds. A major reduction in pathogenic microorganisms is also achieved. Stabilization may be accomplished through a number of methods with anaerobic digestion, the most commonly practiced. Some form of stabilization under either alternative is advocated.

Land disposal is judged to be a viable solution in terms of the above mentioned requirements for both the Atlantic Coast and Great Lakes regions. Suitable areas are available within approximately one hundred miles of all metropolitan regions. The process is particularly appealing as it can allow marginal lands to be upgraded by utilization of the same nutrients which are a problem in inland bodies of water like Lake Erie. Precautions will have to be taken to prevent excess nitrates in ground water, heavy metal build-up in the soil, or transmission of communicable disease agents. However, the operation can be effectively controlled to assure that there will be no environmental degradation by means of seasonal storage, regulated application rates and ground water and surface water monitoring. Sludge pasteurization will not be required unless deemed necessary for obtaining public acceptance of regional waste disposal systems. If this situation occurs, disinfection by heat treatment is the recommended procedure.

The whole subject of ocean disposal of waste materials is still in its infancy. Although increased nourishment to marine life may be a beneficial factor, little basic data is available as to the effect on the ecology of the ocean.

On the basis of limited current information, ocean disposal of digested sludge to the continental shelf area at this time does not appear to be a practical long range solution for the Atlantic coast megalopolis. Although this is judged to be a suitable alternative for Pacific coast communities where relatively great depths of water occur close to shore, the continental shelf on the Atlantic extends up to one hundred miles from shore. The waters of this shelf are not renewed sufficiently to obtain the desirable 3×10^4 dilution in the year 2000 of the total digested plus waste chemical sludge production from the Atlantic metropolitan areas. This dilution requirement is dictated by transparency, nutrient, and dissolved oxygen considerations.

However, although ocean disposal may not be feasible for all communities, it still may be possible for perhaps one-half of the region to utilize this disposal method. Furthermore, by means of new technology, such as selective nutrient removal techniques, it may be feasible to alter the sludge characteristics so as to allow a lower dilution.

Present studies are underway on the effects of waste disposal on ocean biota which should help clarify this situation. Until additional data is available, the only rational approach is to refrain from adding significant additional waste material to the Atlantic shelf.

3.0 WASTE QUANTITIES

3.1 DIGESTED SLUDGE

One result of the operation of municipal wastewater treatment plants is the concentration of pollutional matter into a small portion of the incoming flow. As shown in Figure 3.1.1, among the residues formed are screenings, grit, skimmings, and various forms of sludge. The volume of the first three waste materials is small in comparison with the latter.

Screenings generally range from 0.5 to 5 cu. ft. per million gallons of sewage while grit volumes are normally 1 to 12 cu. ft. per M.G. Screenings are generally high in organic content and are usually buried. Incineration or re-grinding and return of screenings to the waste flow for subsequent removal later in the treatment process is sometimes practiced. Grit is usually of low organic content and is normally used in various forms of land fill applications.

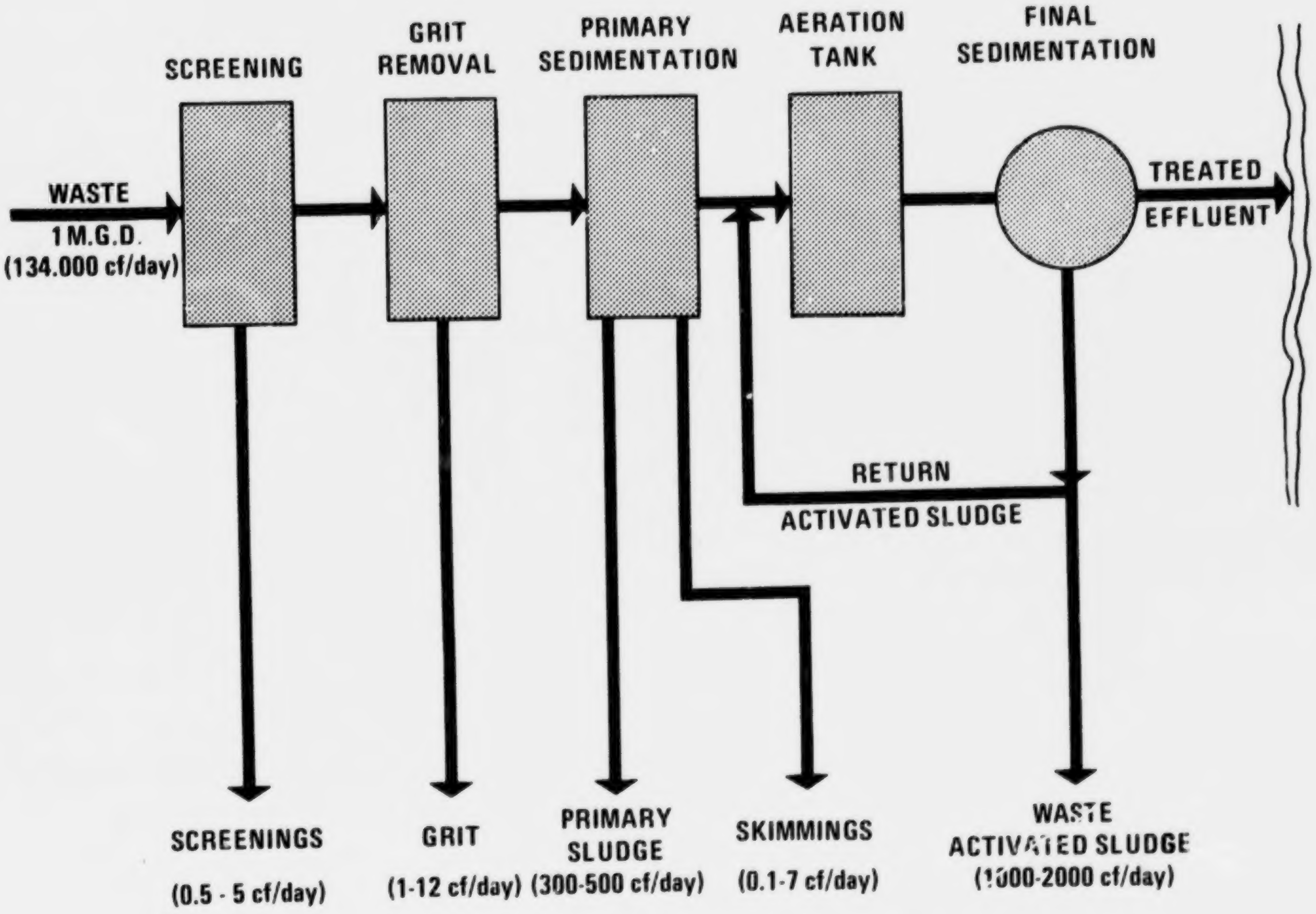
The volume of skimmings normally ranges from 0.1 to 7 cu. ft. per million gallons. In spite of their small volume, disposal has traditionally been a source of difficulty. The material is highly volatile and includes high concentrations of grease and fibrous trash. Among the methods of disposal used are burial, pumping to digesters, dewatering by mechanical equipment and incineration.

Sludge may be defined as the accumulated suspended solids of sewage deposited in tanks or basins. The volume, concentration and general properties are dictated by the raw wastewater characteristics and the method of wastewater treatment used. Raw sludge from primary settling and waste activated sludge from biological treatment are the most common forms of sludge produced in large municipal plants providing secondary treatment. Primary sludge is usually removed at a solids concentration of about 5 percent. It is a highly unstable material and foul in odor. Waste activated sludge is seldom withdrawn at a concentration exceeding 1 percent solids by weight. Concentration by mechanical thickening or be resettling in the primary settling tank is commonly practiced. Unlike primary sludge, fresh waste activated sludge is not odiferous.

To facilitate their ultimate disposal to the environment, waste sludges are frequently treated to reduce their volume or alter their nature. Because sludge contains a significant amount of decomposable material, a form of biological stabilization is often used. Anaerobic digestion has been the most

FIGURE 3.1.1

DAILY RESIDUE PRODUCTION FROM A TYPICAL 1 M.G.D. MUNICIPAL WASTE-WATER TREATMENT PLANT


SCREENING
GRIT REMOVAL
PRIMARY SEDIMENTATION
AERATION TANK
FINAL SEDIMENTATION
WASTE 1 M.G.D. (134.000 cf/day)
TREATED EFFLUENT
RETURN ACTIVATED SLUDGE
SCREENINGS (0.5 - 5 cf/day)
GRIT (1-12 cf/day)
PRIMARY SLUDGE (300-500 cf/day)
SKIMMINGS (0.1-7 cf/day)
WASTE ACTIVATED SLUDGE (1000-2000 cf/day)

popular and widely practiced method of sludge stabilization for many years. However, aerobic digestion has recently received increased attention due to the rapid growth of "extended aeration" treatment plants. Characteristics of anaerobically digested sludge are given in Appendix A.

In this section, digested sludge quantities are projected for metropolitan areas in two broad geographical regions, namely the Great Lakes region from Buffalo to Milwaukee and the Atlantic Coast region from Boston to Norfolk. In lieu of a detailed analysis of current and future sludge loads by areas in each region, a generalized method was used. This method is based upon determining typical values of sewage composition and similarly, typical effects of wastewater treatment. Digested sludge production is taken as the product of suspended solids contribution, removal of suspended solids in the treatment process, digestion loss, and population served. Although sludge quantities for individual communities may deviate appreciably from the average values given here, the method is felt to be valid for determining the approximate total amount of digested sludge in each region.

3.1.1 Suspended Solids Contribution

In any community, the wastes discharged from different sources may vary widely in composition and concentration. However, the elaborate sewage collection system of most communities results in the delivery of a relatively homogenous product to the treatment plant. This is particularly true if monthly average suspended solids or biochemical oxygen demand is considered.

Many design manuals and textbooks have indicated that an acceptable per capita value for the design of municipal wastewater treatment facilities is 0.20 lb/day of suspended solids. Since this factor is basic to determining sludge volumes, its validity is of prime interest.

Recently Loehr (1968) made a study on the variation of wastewater quality parameters throughout the United States. Data were obtained on a total of 73 cities in 27 states for the period 1958 through 1964. His results showed the following average per capita loadings:

Sample	Number of Cities	Suspended Solids lb/cap/day
All cities	73	0.23
Cities over 50,000 Population	38	0.25

The entire range for all cities sampled was 0.08 to 0.73 lb/cap/day for suspended solids. Wastewater flows averaged 135 gpcd.

The higher per capita suspended solids loadings are not surprising. Industrial activity is reflected in these values, particularly for cities with a population of 50,000 or more. These industrial wastes may more than double the suspended solids contribution calculated on a domestic population basis. For example, Metropolitan Chicago has adopted a program to provide a high level of treatment for all liquid wastes, domestic, commercial, or industrial, that occur within its area of jurisdiction. As a result, Chicago reports about 0.4 lb/cap/day of suspended solids (Harza Engineering Co., 1967). For design purposes an adequate study of the wastewater characteristics of a community must be made rather than relying on average per capita data. However, for projecting sludge volumes over broad regions, the average values are still useful.

An additional factor is the use of home garbage grinders. Only in recent years has this become significant. The American Society of Civil Engineers (1959) estimated that garbage grinding could increase suspended solids by an average of 60%. In a study in Aurora, Colorado (Watson and Clarke, 1961), it was found that when 58% of households had garbage grinders, suspended solids per capita increased by 0.064 lbs/day. The above indicates that garbage grinding may result in increasing suspended solids by about 0.12 lbs/cap/day.

Since the primary concern of this study is with large metropolitan areas, a value of 0.25 lb/cap/day was taken as the total suspended solids contribution in 1961, the midyear of Loehr's data. Garbage grinding was used by about 12% of the U. S. population at that time. Therefore, a 1961 base load of 0.24 lbs/cap/day and a garbage grinding addition of 0.01 lbs/cap/day was assumed. In 1968, about 15 to 16% of the population used garbage grinders, resulting in an addition of 0.02 lb/cap/day. For the years 1975 and 2000, contributing percentages of 25% and 40% respectively were projected. This would result in garbage grinding loads of 0.03 lb/cap/day in 1975 and 0.05 lb/cap/day in the year 2000. If it is assumed that the increase of suspended solids from industrial wastes discharged to the public sewers will be proportional to population growth, the base load can be expected to remain constant. Therefore, the following suspended solids projections were obtained.

Year	Base Load lbs s. s. /cap/day	Garbage Grinding lbs s. s. /cap/day	Total lbs s. s. /cap/day
1961	0.24	0.01	0.25
1968	"	0.02	0.26
1975	"	0.03	0.27
2000	"	0.05	0.29

3.1.2 Fresh Sludge Production

In wastewater treatment plants, a number of unit operations are applied in various combinations to meet particular disposal criteria. Primary treatment generally refers to plain sedimentation while secondary treatment normally means biological treatment as represented by trickling filtration or the activated sludge process. In terms of suspended solids removal, the approximate efficiency of these various operations is as follows (Fair and Geyer, 1954):

Operation or Process	% Removal of Suspended Solids
Plain sedimentation	40 - 70
Trickling filtration preceded and followed by sedimentation	70 - 92
Activated sludge treatment preceded and followed by sedimentation	85 - 95

Over the last two decades, there has been rapid growth in municipal waste treatment capabilities in the United States. Both the number of treatment plants and the average level of treatment have risen steadily. This may be seen from the table below (F. W. P. C. A., 1968a).

Estimated Population Served (Millions of People)	1949	1957	1962
Sewers alone	28.1	21.9	14.7
Primary treatment	17.2	25.7	32.7
Secondary treatment	26.1	43.3	61.2

For the plants in the Atlantic and Great Lakes regions in 1968, it was estimated that about 80% of the population in each region was served by secondary treatment facilities and most of the remainder by primary facilities. Furthermore, with the passage of the Water Pollution Control Act of 1965 and the Clean Water Restoration Act of 1966, it is clear that a commitment has been made to expedite the continual upgrading of municipal waste treatment facilities. By 1975, secondary treatment should become the norm. For most metropolitan areas, this will probably mean increased use of activated sludge treatment which is the most widely used form of secondary treatment today.

For plain sedimentation alone, the amount of fresh sludge produced will be taken as equal to 60% of the incoming suspended solids. From the suspended solids projection given in the preceding section, this results in 0.156 lbs/cap/day for primary treatment in the year 1968.

In the activated sludge process, solids are produced through biological synthesis as well as destroyed through endogenous respiration. The quantity of waste activated sludge produced is dependent upon the type of waste handled as well as how the process is operated. However, as evidenced by various plant operating records, the total amount of primary and waste activated sludge sent to the digesters is often only somewhat higher than the reported suspended solids removed. Furthermore, because many plants recycle either digester supernatant, elutriate or filtrate, or a combination thereof, these records overstate the amount of fresh sludge produced. Therefore, as an approximation over a broad region, total fresh sludge production with secondary treatment will be taken equal to the amount of suspended solids removed. Assuming 90% removal, secondary sludge production for 1968 and 1975 will equal 0.234 and 0. 243lbs/cap/day respectively.

3.1.3 Tertiary Treatment

By the year 1985, one may expect that secondary treatment will not suffice in many areas. Indeed even now, a number of areas, particularly in the Great Lakes region, are planning for tertiary treatment. In this report, tertiary treatment is defined as further purification beyond conventional secondary treatment or a modification or replacement of one or more steps comprising present conventional treatment which results in the removal of additional contaminants. The type of treatment is dictated by the specific use intended for the waste stream. In some cases, the goal may be to obtain further reduction in BOD and suspended solids while in others, nutrient removal may take priority. Many physical-chemical separation processes are under examination, including absorption, distillation, electrodialysis, reverse osmosis, and ion exchange. More conventional wastewater treatment methods such as sand filtration, micro straining and particularly chemical clarification are being given strong consideration.

Considering the different goals for tertiary treatment and the many possible methods available for obtaining these goals, a variety of effects may result from this step. However, it is likely that in all cases, suspended solids removal will increase. Furthermore, since phosphate removal is currently advocated and chemical precipitation can effectively remove phosphates as well as suspended solids (F. W. P. C. A., 1968b), a significant amount of waste chemical sludge may be produced.

A number of investigations for phosphate removal with lime or alum have indicated that chemical dosages up to several hundred ppm may be required (F. W. P. C. A., 1968b). For suspended solids removal alone, required dosages appear to be somewhat lower. For this study, it has been assumed that all suspended solids from the secondary process are removed and show up as additional fresh sludge. This results in a total projected fresh sludge load of 0.29 lb/cap/day. Furthermore, it is assumed that an equivalent chemical dosage of 50 ppm will end up as sludge for disposal. Conceivably this increment may be much higher, but the present state of the art leaves much room for possible improvement. Using an assumed average flow of 175 gal/cap/day, we arrive at a waste chemical sludge load of 0.07 lb/cap/day for tertiary treatment in the year 2000.

3.1.4 Digestion Losses

Anaerobic digestion results in achieving a substantial degree of sludge stabilization. A secondary benefit is a reduction in the amount of solids for disposal. The solids in combined sludges from primary and secondary treatment are approximately 70 percent volatile. Digestion commonly results in the destruction of about 60 percent of these volatile solids. For the sludge projection in this section, a 40% reduction in the weight of fresh sludge has been taken.

3.1.5 Standard Metropolitan Statistical Areas

For identifying the sludge sources in the two study regions, Standard Metropolitan Statistical Areas (S. M. S. A's) were selected. This grouping was particularly suited for this study because it covers all large metropolitan areas for which population data is readily available. Citing from the 1968 Statistical Abstract of the United States, the S. M. S. A. includes the county in which the central city is located and adjacent counties that are found to be metropolitan in character and economically and socially integrated with the county of the central city. An exception is made in New England in that towns comprise the units rather than counties. An S. M. S. A. may cross state lines. Each S. M. S. A. contains at least:

a) One central city with 50,000 inhabitants or more or
b) Two cities having continuous boundaries and constituting, for general economic and social purposes, a single community with a combined population of at least 50,000, the smaller of which must have a population of at least 15,000.

Figures 3.1.2 and 3.1.3 show the S. M. S. A.'s contained with the Great Lakes and Atlantic Coast study regions.

3.1.6 Population Projections

Populations for each S. M. S. A. in the two study regions were obtained for 1965 or 1966 from Bureau of the Census reports (Series P-25, No.'s 371 & 378). For projecting populations to the year 2000, the rate of growth for each S. M. S. A. was assumed to be the same as the rate of growth for the containing state or states. State population increases up to 1985 were also based on Census reports (Series P-25, No. 375). The Series II-D rate of growth was used. This implies that interstate migration rates will change from recent levels so as to result in no net migration among states on 50 years and that there will be a continued decline from present national fertility levels. The recently reported 17.4 births per 1,000 population in 1968, which was the

Figure 3.1.2

F.W.P.C.A. Waste Management Study

GREAT LAKES MEGALOPOLIS WITH LOCATION OF STANDARD METROPOLITAN STATISTICAL AREAS

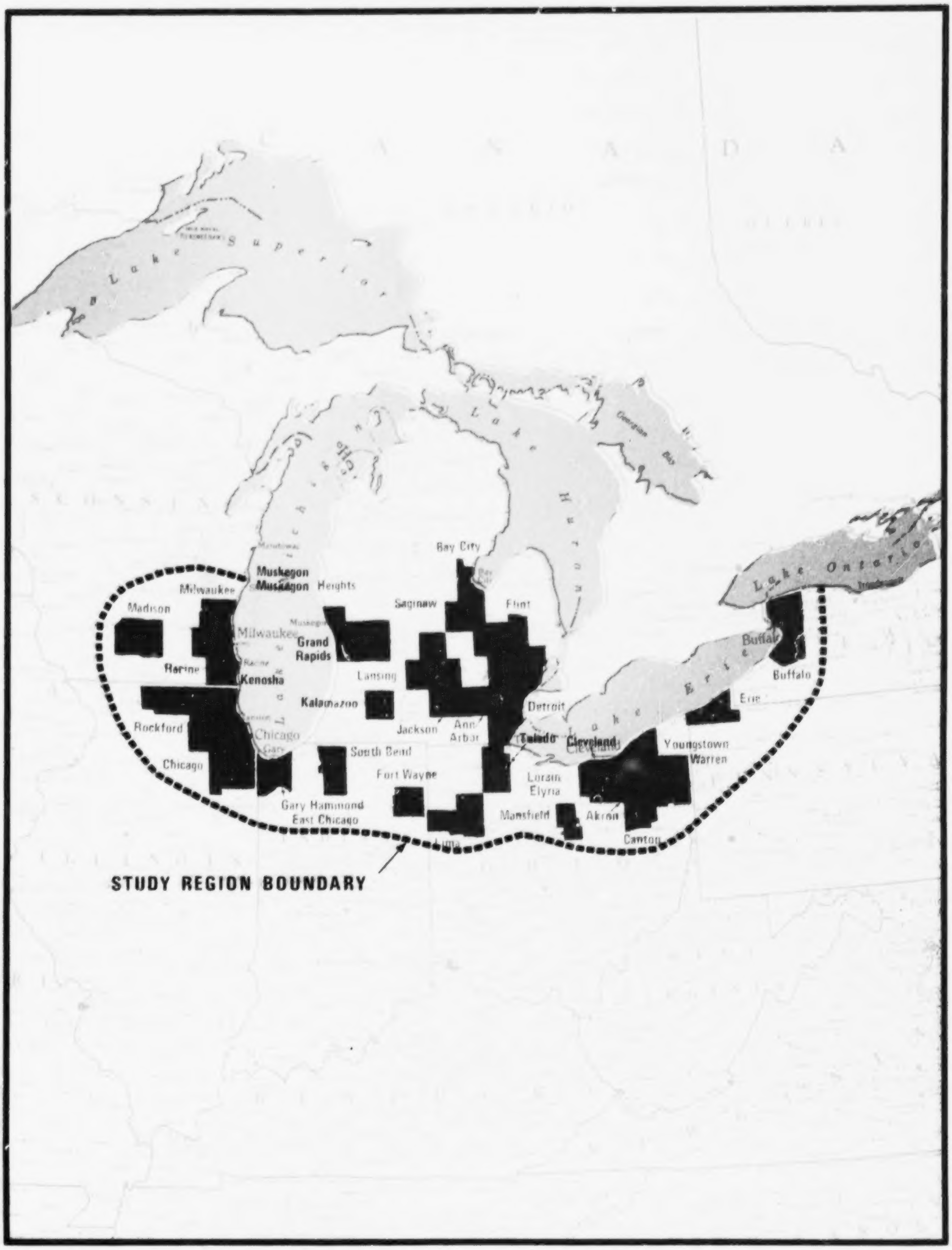
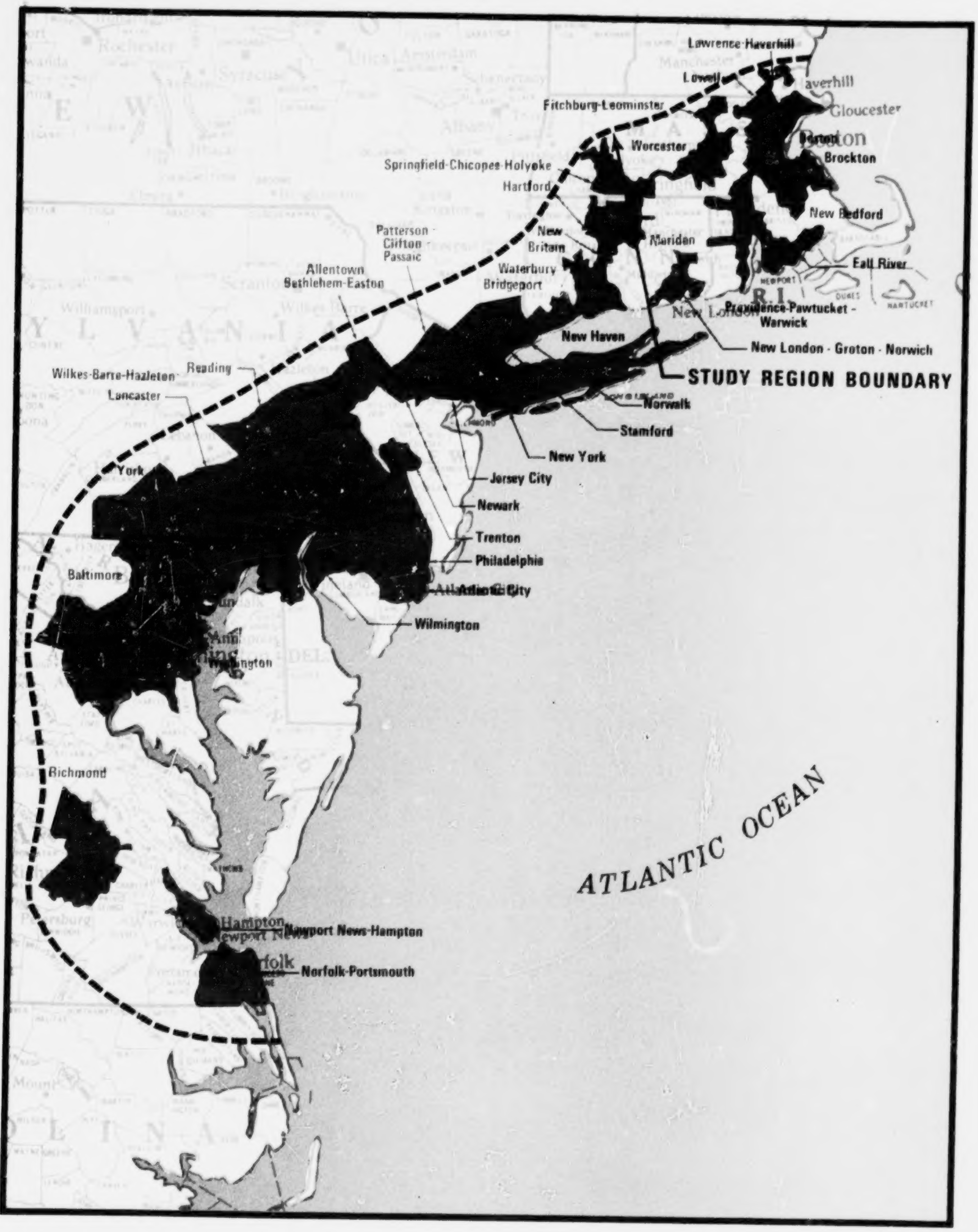

CANADA
Lake Superior
Lake Michigan
Lake Huron
Georgian Bay
Lake Ontario
Lake Erie
Bay City
Muskegon
Muskegon Heights
Milwaukee
Madison
Saginaw
Flint
Grand Rapids
Racine
Kenosha
Lansing
Kalamazoo
Rockford
Chicago
South Bend
Jackson
Ann Arbor
Detroit
Toledo
Cleveland
Fort Wayne
Gary Hammond
East Chicago
Lima
Lorain
Elyria
Mansfield
Akron
Canton
Youngstown
Warren
Erie
Buffalo
STUDY REGION BOUNDARY

Figure 3.1.3

ATLANTIC COAST MEGALOPOLIS
WITH LOCATION OF STANDARD METROPOLITAN STATISTICAL AREAS


Lawrence-Haverhill
Lowell
Haverhill
Gloucester
Fitchburg-Leominster
Worcester
Boston
Brockton
Springfield-Chicopee-Holyoke
Hartford
New Bedford
Patterson-Clifton-Passaic
New Britain
Meriden
Fall River
Allentown Bethlehem-Easton
Waterbury
Bridgeport
Providence-Pawtucket-Warwick
New Haven
New London - Groton - Norwich
STUDY REGION BOUNDARY
Wilkes-Barre-Hazleton
Reading
Lancaster
Norwalk
Stamford
New York
York
Jersey City
Newark
Trenton
Philadelphia
Baltimore
Atlantic City
Wilmington
Washington
Richmond
Newport News-Hampton
Norfolk-Portsmouth
ATLANTIC OCEAN

lowest birth rate since 1940, is in agreement with the second assumption. For the period 1985 through 2000, the 1975 - 1985 rate of growth was assumed applicable.

3.1.7 Sewered Populations

In 1968, the urban population in the United States was approximately 146 million people (FWPCA, 1968a). Wastes from about 32 million, or approximately 22 percent of the urban population, were not given any treatment. By the end of fiscal year 1973, adequate treatment is planned to be achieved for the total urban population. This is consistent with implementation plans of most approved state standards. A secondary type of waste treatment is generally taken as representative of that contained in the standards.

However, the S. M. S. A.'s do not include only urban population. Outlying portions of the surrounding counties to the central city contain some rural areas. Therefore, a derating factor must be applied to the total population in each S. M. S. A. to account for the rural dwellers who will continue to use small package plants or septic tanks. As time passes, population increases will result in shrinking the already small amount of rural areas in each S. M. S. A. and the sewered population will approach the total population. In this study, it has been assumed that for 1968 and 1975, 90 and 95 percent, respectively, of the total population in each S. M. S. A. will be sewered. For the year 2000, a 98 percent factor has been taken.

3.1.8 Projected Sludge Quantities

In Tables 3.1.1 and 3.1.2 each S. M. S. A. has been listed for the Great Lakes and Atlantic Coast study regions. Populations, as discussed above, have been projected for 1968, 1975 and the year 2000. They indicate that the population of the Great Lakes Megalopolis which was 22.9 million in 1968 will increase by 5% to 24.2 million in 1975 and by 37% to 31.4 million in the year 2000. In the Atlantic Coast Megalopolis, a 7% increase from the 37.2 million population in 1968 to 39.6 million in 1975 is indicated. In the year 2000, a 38% increase to 51.2 million is expected.

To determine the digested sludge quantity in 1968, the total population

TABLE 3.1.1

GREAT LAKES STUDY REGION

PROJECTED POPULATIONS

S. M. S. A.	Population in Millions		
	1968	1975	2000
Buffalo, N. Y.	1.35	1.43	1.79
Erie, Pa.	.26	.26	.30
Cleveland, Ohio	2.04	2.16	2.84
Akron, Ohio	.66	.70	.92
Canton, Ohio	.36	.39	.51
Youngstown-Warren, Ohio	.53	.57	.74
Toledo, Ohio	.67	.72	.94
Lorain-Elyria, Ohio	.25	.26	.34
Mansfield, Ohio	.13	.14	.18
Lima, Ohio	.11	.12	.16
Detroit, Mich.	4.12	4.34	5.57
Ann Arbor, Mich.	.19	.20	.26
Flint, Mich.	.48	.50	.64
Saginaw, Mich.	.21	.22	.29
Bay City, Mich.	.11	.12	.15
Lansing, Mich.	.34	.36	.46
Jackson, Mich.	.14	.15	.19
Muskegon-Muskegon Hts., Mich.	.16	.16	.21
Grand Rapids, Mich.	.51	.54	.69
Kalamazoo, Mich.	.18	.19	.25
So. Bend, Ind.	.28	.29	.37
Ft. Wayne, Ind.	.26	.28	.36
Gary-Hammond-E. Chicago, Ind.	.61	.64	.82
Chicago, Ill.	6.83	7.21	9.48
Rockford, Ill.	.25	.27	.35
Kenosha, Wisc.	.12	.12	.16
Racine, Wisc.	.16	.17	.22
Milwaukee, Wisc.	1.35	1.41	1.86
Madison, Wisc.	.26	.27	.36
Total Population in millions	22.92	24.19	31.41
Percent Increase Over 1968	-	5	37

TABLE 3.1.2

ATLANTIC COAST STUDY REGION

PROJECTED POPULATIONS

	Population in Millions		
S. M. S. A.	1968	1975	2000
Norfolk-Portsmouth, Va.	.65	.71	.96
Newport News-Hampton, Va.	.28	.31	.42
Richmond, Va.	.51	.55	.75
Washington, D. C.	2.67	2.90	4.21
York, Pa.	.31	.31	.37
Lancaster, Pa.	.29	.30	.34
Baltimore, Md.	2.04	2.28	3.31
Wilmington, Del. - N. J. - Md.	.48	.54	.78
Reading, Pa.	.28	.29	.34
Philadelphia, Pa.	4.71	4.80	5.58
Allentown-Bethlehem-Easton, Pa.	.52	.53	.61
Atlantic City, N. J.	.18	.20	.28
Trenton, N. J.	.31	.34	.47
Newark, N. J.	1.91	2.11	2.91
Jersey City, N. J.	.63	.70	.96
New York, N. Y.	11.62	12.37	15.47
Paterson-Clifton-Passaic, N. J.	1.36	1.49	2.06
Bridgeport-Stamfort-Norwalk, Conn.	.78	.86	1.18
New Haven, Waterbury, Meriden, Conn.	.73	.81	1.12
New London, Groton, Norwich, Conn.	.23	.25	.34
Hartford-New Britain, Conn.	.79	.87	1.20
Providence, Pawtucket, Warwick, R. I.	.74	.77	.89
New Bedford, Fall River, Mass.	.42	.43	.50
Boston, Lawrence, Haverhill, Lowell, Mass.	3.23	3.37	4.21
Brockton, Mass.	.30	.31	.39
Worcester, Fitchburg, Leominster, Mass.	.62	.64	.80
Springfield-Chicopee-Holyoke, Mass.	.56	.58	.73
Total Population in millions	37.15	39.62	51.18
Percent Increase Over 1968	-	7	38

in each study region was first multiplied by the sewered population factor. As shown in Table 3.1.3, this population served was then subdivided into two fractions, namely, the 80% undergoing secondary treatment and the 20% with primary treatment. Each fraction was multiplied by the respective per capita fresh solids production to determine the proportional amount of fresh sludge from each process. The sum was multiplied by 0.60 to account for the loss in digestion. The product is the total digested sludge that would be currently produced under the assumed uniform sewage and treatment processes.

A similar procedure was followed for the year 1975 except all sewage was assumed to have secondary treatment. Likewise, a calculation was performed for the year 2000, assuming tertiary treatment. A separate determination of the additional waste chemical sludge that may result from teritary treatment is also shown.

As shown in the table, sludge volumes in both the Great Lakes region and Atlantic Coast region are expected to significantly increase in the next thirty-two years. In 1968, digested sludge volumes are estimated at 1350 and 2190 tons/day respectively. By the year 1975, this is projected to increase to 1670 tons/day in the Great Lakes Area and to 2740 tons/day in the Atlantic Coast region. For the year 2000, total digested and waste chemical sludge volumes may total 3760 tons/day and 6120 tons/day respectively. In terms of percent increases over 1968, sludge volumes will increase by nearly 25% to 1975 and 180% to the year 2000 in both study regions. It is evident that sludge disposal will become a much larger problem in the near future.

TABLE 3.1.3

PROJECTED DIGESTED SLUDGE QUANTITIES

	Great Lakes			Atlantic Coast		
Year	1968	1975	2000	1968	1975	2000
Total Population (millions)	22.92	24.19	31.41	37.15	39.62	51.18
% Sewered	90	95	98	90	95	98
Sewered Population (millions)	20.63	22.98	30.78	33.44	37.64	50.16
Primary Treatment						
% of Sewered Pop'n served	20	-	-	20	-	-
Population served (millions)	4.13	-	-	6.69	-	-
Fresh sludge production (lbs/cap/day)	.156	-	-	.156	-	-
Primary fresh sludge (dry tons/day)	320	-	-	520	-	-
Secondary Treatment						
% of Sewered Pop'n served	80	100	-	80	100	-
Population served (millions)	16.50	22.98	-	26.75	37.64	-
Fresh sludge production (lbs/cap/day)	.234	.243	-	.234	.243	-
Fresh sludge (dry tons/day)	1930	2790	-	3130	4570	-
Tertiary Treatment						
% of Sewered Pop'n served	-	-	100	-	-	100
Population served (millions)	-	-	30.78	-	-	50.16
Fresh sludge production (lbs/cap/day)	-	-	.29	-	-	.29
Fresh sludge (dry tons day)	-	-	4460	-	-	7270
Waste chemical sludge prod. (lbs/cap/day)	-	-	.07	-	-	.07
Waste chemical sludge (dry tons/day)	-	-	1080	-	-	1760
Total fresh sludge (dry tons day)	2250	2790	4460	3650	4570	7270
Digestion loss factor	.60	.60	.60	.60	.60	.60
Total digested sludge (dry tons/day)	1350	1670	2680	2190	2740	4360
Total waste chemical sludge (dry tons/day)	-	-	1080	-	-	1760
Total digested & waste chemical sludge (dry tons/day)	1350	1670	3760	2190	2740	6120
(dry tons/year x 10^3)	490	610	1370	800	1000	2230
Percent increase from 1968	-	24	178	-	25	179

3.2 MAINTENANCE DREDGINGS

The Commerce Clause of the Constitution, which gave Congress the power to regulate commerce with foreign nations and among the various states, has been the basic authority for the participation of the Corps of Engineers in the development of water resources. In 1824, Congress directed the U. S. Army Corps of Engineers to study and plan a national transportation system including waterways. Since that time, the Corps has been responsible for the development of the harbors and rivers of the U. S. Economic feasibility studies are made by the Corps for all projects. Authorization is approved by both houses of Congress and appropriations are provided in the annual budget.

Dredging of the channels and harbors insures safe and efficient movement of traffic and increases the economic potential of the waterfront lands. Two general classifications of dredging may be made, namely maintenance dredging and new work.

Maintenance dredging is the removal of relatively thin, poorly consolidated deposits in established harbors and channels, on a scheduled basis. New work is the removal of thicker layers of more consolidated material, to enlarge or deepen the existing facilities or to develop new facilities. The latter is frequently virgin material and is only slightly contaminated. On the other hand, maintenance dredgings are comprised of the recently deposited material and therefore are more apt to contain pollutants. This report considers only the Corps of Engineers maintenance dredgings. Characteristics are given in Appendix A.

3.2.1 Historical Record

Great Lakes Dredging

The Great Lakes study area includes the lakefront of the states of Pennsylvania, Ohio, Michigan and Indiana. Also included is the Lake Michigan portion of Wisconsin from Milwaukee south to the state line and the Lake Erie portion of New York from Niagara Falls west and south to the state line. The North Central Division of the Corps of Engineers, through its district offices at Chicago, Detroit, Buffalo and St. Paul, has the responsibility for all of the dredging programs in the study area. All districts do as much of their own maintenance dredging as possible, but use private contractors for projects that do not lend themselves to Corps of Engineers equipment.

In the Great Lakes, the Corps of Engineers has established shoaling rates for each project. This is the average amount of suspended matter that will be deposited each year by the river or stream. The maintenance dredging work conforms approximately to these values and has been taken as equivalent. The total quantity by state is given in Table 3. 2. 1. Assuming a cubic yard of dredgings contains 1250 pounds of dry solids (U. S. Army Corps of Engineers, 1969) maintenance dredging quantities are shown in cubic yards as well as tons of dry solids. In Table 3. 2. 2 the dredging frequency and maintenance work is listed by project.

Atlantic Coast Dredging

The limits of the Atlantic Coast study area are from the north boundary of Massachusetts to the southern boundary of Virginia. This includes all river and harbor projects of the North Atlantic Division of the Corps of Engineers and part of the New England Division. The North Atlantic Division operates through district offices at New York, Baltimore, Philadelphia and Norfolk. The New England Division operates through a central office near Boston.

Shoaling rates have not been established for the projects in the Atlantic Coast area. However, by reviewing the Chief of Engineer's Annual Reports covering the five years 1962-1966, the recent annual credited workload of each district was established. This credited volume corresponds to the desired removal as shown by "neat lines" on plans. Since it is necessary to dredge beyond the neat lines to insure complete removal, the bin measurement or actual amount hauled exceeds this credited volume. For the years 1962 through 1966, the bin measured volume of all dredging averaged about 30% greater. Using this basis and a reassumed 1250 pounds of dry solids per c.y., average dredging volumes are summarized in Table 3. 2. 3 by district for the Atlantic Coast area.

TABLE 3.2.1

GREAT LAKES STUDY AREA
MAINTENANCE DREDGING QUANTITIES

SUMMARY SHEET

State	Equivalent Annual Maintenance Dredging Quantities[(a)] Cubic Yards	Equivalent Annual Maintenance Dredging Quantities[(a)] Tons of Dry Solids
NEW YORK	647,000	404,400
PENNSYLVANIA	300,000	187,500
OHIO	4,500,000	2,812,500
INDIANA	392,000	245,100
ILLINOIS	34,000	21,300
MICHIGAN	3,314,500	2,071,800
WISCONSIN	582,400	365,000
TOTAL:	9,769,900	6,107,600

(a) Based on 1968 Shoaling Rates established by the Corps of Engineers.

TABLE 3.2.2

GREAT LAKES STUDY AREA

MAINTENANCE DREDGING QUANTITIES

Project Location	Dredging Frequency	Equivalent Annual Maintenance Dredging Quantities[(a)] Cubic Yards	Tons of dry solids
NEW YORK			
Dunkirk Harbor	Annual	20,000	12,500
Buffalo Harbor	Annual	400,000	250,000
Buffalo (Ship Canal)	Annual	125,000	78,100
Black Rock Channel & Tonawanda Harbor	Annual	100,000	62,500
Little River (Niagara)	10 Yrs.	2,000	1,300
PENNSYLVANIA			
Erie Harbor	Annual	300,000	187,500
OHIO			
Toledo Harbor	Annual	1,400,000	875,000
Sandusky Harbor	Annual	600,000	375,000
Huron Harbor	Annual	200,000	125,000
Lorain Harbor	Annual	300,000	187,500
Cleveland (Cuyahoga River)	Annual	720,000	450,000
Cleveland Harbor	Annual	560,000	350,000
Fairport Harbor	Annual	400,000	250,000
Ashtabula Harbor	Annual	220,000	137,500
Conneaut Harbor	Annual	100,000	62,500
INDIANA			
Indiana Harbor	Annual	200,000	125,000
Michigan City Harbor	Annual	46,000	28,800
Calumet Harbor & River	Annual	146,000	91,300
ILLINOIS			
Waukegan Harbor	Annual	34,000	21,300

GREAT LAKES STUDY AREA (cont.)

Project Location	Dredging Frequency	Equivalent Annual Maintenance Dredging Quantities[(a)]	
		Cubic Yards	Tons of dry solids
MICHIGAN			
Ontonagon Harbor	Annual	80,000	50,000
Keweenaw Waterway	3-4 Yrs.	40,000	25,000
Big Bay Harbor	Annual	5,000	3,100
St. Mary's River	Annual	250,000	156,300
Au Sable Harbor	Annual	40,000	25,000
Saginaw River	Annual	600,000	375,000
Sebewaing River	4 Yrs.	17,000	10,600
Harbor Beach Harbor	3 Yrs.	35,000	21,900
St. Clair River	Annual	200,000	125,000
Rouge River	Annual	300,000	187,500
Detroit River	Annual	850,000	531,300
St. Joseph Harbor	Annual	100,000	62,500
So. Haven Harbor	Annual	80,000	50,000
Saugatuck Harbor	Annual	55,000	34,400
Holland Harbor	Annual	100,000	62,500
Grand Haven Harbor	Annual	100,000	62,500
Muskegon Harbor	Annual	105,000	65,600
White Lake Harbor	Annual	60,000	37,500
Pentwater Harbor	Annual	70,000	43,800
Ludington Harbor	Annual	55,000	34,400
Manistee Harbor	Annual	55,000	34,400
Portage Lake Harbor	2 Yrs.	15,000	9,400
Frankfort Harbor	Annual	32,500	20,300
Charlevoix Harbor	Annual	30,000	18,800
Manistique Harbor	Annual	40,000	25,000
WISCONSIN			
Duluth-Superior Harbor	Annual	150,000	93,800
Menominee Harbor	3 Yrs.	8,000	5,000
Pensaukee Harbor	3-5 Yrs.	2,100	1,300
Oconto Harbor	5 Yrs.	3,000	1,900
Green Bay Harbor	Annual	121,000	75,600
Sturgeon Bay	Annual	50,000	31,300
Kewaunee Harbor	Annual	33,000	21,600
Two Rivers Harbor	Annual	52,000	32,500
Manitowoc Harbor	Annual	40,000	25,000
Sheboygan Harbor	Annual	20,000	12,500
Port Washington Harbor	Annual	8,000	5,000
Milwaukee Harbor	Annual	48,000	30,000
Racine Harbor	Annual	21,500	13,400
Kenosha Harbor	Annual	25,800	16,100

(a) Based on 1968 Shoaling Rates established by the Corps of Engineers.

TABLE 3. 2. 3

ATLANTIC COAST STUDY AREA
MAINTENANCE DREDGING QUANTITIES

District	Average Credited Volume in C. Y. /yr.(b)	Actual Material Removed	
		C. Y. /yr.(d)	Dry Tons/yr.
BALTIMORE	1,078,000 (c)	1,401,000	876,000
NORFOLK	4,200,000	5,460,000	3,413,000
PHILADELPHIA	12,530,000	16,290,000	10,181,000
NEW ENGLAND (a)	1,188,000	1,544,000	1,965,000
NEW YORK	3,081,000	4,005,000	2,503,000
TOTAL:	22,077,000	28,700,000	17,938,000

(a) Includes work in Massachusetts, Connecticut, and Rhode Island only.

(b) Based on records for years 1962 through 1966.

(c) Includes average Ocean City dredgings based on the years 1963 through 1966 only.

(d) Assuming a uniform 30% increase over the credited volume.

3.2.2 Projected Waste Quantities

Great Lakes Projected Quantities

Although future increases in water-borne commerce for the Great Lakes region can be anticipated, projected maintenance dredging volumes are not expected to increase above the present level of operations. The basis for this forecast is a March 1969 study prepared by the U. S. Army Corps of Engineers entitled "Dredging and Water Quality Problems in the Great Lakes". The following statements summarize the reasons for this position.

It is unlikely that further deepening of harbors and connecting channels by new work dredging will occur in the near future. Almost all major harbors have been improved to accomodate the size of vessel which can pass through the ship locks on the St. Lawrence River, the Welland Canal, or the St. Mary's River. Costs for further improvement by new work dredging would be of such magnitude that it is doubtful whether such improvements could be justified.

There are two factors which could somewhat reduce the future volume of maintenance dredging. One is the possibility of increased soil conservation practices. To date, such practices have been implemented only at small watersheds. The second factor is the current effort toward reduction of municipal and industrial waste discharges. However, these wastes comprise only a minor portion of the total volume of maintenance dredging.

Atlantic Coast Projected Quantities

On the East Coast, a somewhat similar situation with respect to future maintenance dredging exists. It is questionable whether many Atlantic Coast harbors will be subject to further deepening in the future. In a report entitled "Harbor and Port Development" by the U. S. Army Corps of Engineers (1968) several restraints are given. A number of Atlantic harbors already face one or more of these potential obstacles:

1. Significant costs may result from removal and relocation of industrial, commercial and residential structures due to related channel widening.
2. Further deepening may require expensive rock removal or long extensions of approach channels.
3. Acceptable spoil disposal sites may become increasingly difficult to obtain.

4. Navigation improvements may adversely affect the marine ecology by altering bottom conditions or the salt-fresh water interface.

5. Further deepening may entail danger to groundwater supplies if impervious layers of protective rack are damaged so that salt water is permitted to seep into the acquifers.

In reviewing the Chief of Engineer's Annual Reports for the years 1962 through 1966, no discernable trend in reported maintenance dredging work in the Atlantic Coast region was observed. Although an analysis of future dredging expectations at individual harbors is beyond the scope of this study, the restraints mentioned above lead one to suspect that future increases in maintenance dredging may be minimal. Based on the general trend of total new plus maintenance dredging work over the last fifteen years, a 1% increase per year for maintenance dredging work in the Atlantic Coast region has been taken for projecting future work. As shown in Table 3. 2. 3, 17. 9 million tons were assigned to the year 1964. On this basis, waste quantities of approximately 18. 6, 20. 0 and 24. 6 million tons are projected for the years 1968, 1975 and 2000 respectively.

3.3 FLY ASH

The term fly ash, as used in this study, refers to the particulate residue which results from the burning of pulverized coal. Fly ash is also produced from other fuels, such as residual fuel oil. However, these quantities are small compared to those produced from coal burning, and are not considered. Characteristics of fly ash are given in Appendix A.

In 1965, coal consumption in the U. S. A. was approximately 450 million tons, usage being divided as follows (Brackett, 1967):

Electric Utilities	53%
Industrial Uses of Coal	22%
Coke Products and Minor Uses	25%

This consumption of coal produced 20 million tons of fly ash, of which 0.6 tons or 3% was utilized in other industries. The remaining fly ash was disposed of as a solid waste. It can be seen from the details given above that electric power generation plants are the major source of this waste material. This study considers only the power generation portion of the total fly ash. In metropolitan areas, this source of fly ash could result in disposal problems as the amount produced by any one plant can be relatively large.

The collection of fly ash is accomplished by either mechanical or electrostatic methods. Mechanical collection is achieved by cyclones or centrifugal collectors, and maximum efficiencies of 80 - 85 percent are common with this type of equipment. Electrostatic collection consists of subjecting ash-laden stack gases to a high voltage DC field, which charges the particles, and then depositing the particles on charged collection plates by electrostatic forces. The efficiency of this type of separation can be in excess of 99%. Current use tends to favor electrostatic separation and as air pollution control legislation becomes more stringent, greater use is expected.

3.3.1 Projected Coal Burn

A number of electric power projections were reviewed to determine the expected increase in coal consumption during the study period. They indicated that the national coal consumption in 1975, 1985, and 2000 could be 130%, 170% and 110%, respectively, of 1966-67 use. The expected fall-off from 1985 to the year 2000 is due primarily to projected nuclear power growth. Also, as air pollution control becomes more rigid, some coal is expected to be displaced by low sulphur oil.

Tables 3.3.1 and 3.3.2 list current power plant coal burn tonnages for various areas in the Great Lakes and Atlantic Coast study regions. Projected coal burn through the year 2000 is obtained from these tables and the percent increases indicated below:

COAL BURN
(Million Tons/Year)

Year	% of 1966-67	Atlantic Coast Region	Great Lakes Region
1966-67	100	35.9	46.1
1975	130	46.7	59.9
1985	170	61.0	78.4
2000	110	39.5	50.7

3.3.2 Utilization

In terms of percentage used, the greatest success in fly ash utilization has been in Europe. This is evident from the 1966 ash utilization data for various countries given in Table 3.3.3. Although this table includes bottom ash and boiler slag, it still is indicative of the big difference between European and United States fly ash utilization.

The European success is a result of economic and environmental pressures. For example, product prices in France are controlled by the Public Authorities. The cement manufacturers found that authorized price increases did not keep pace with their increasing production costs. Investigation revealed that production could be economically increased by inclusion of fly ash in the final product. With fly ash source areas in close proximity to major cement producers, utilization was achieved (Jarrige, 1967).

TABLE 3.3.1

GREAT LAKES STUDY AREA

CURRENT POWER PLANT COAL BURN AND ESTIMATED FLY ASH PRODUCTION (1)

	Coal Burn (tons/yr)	Fly Ash Production (tons/yr) (2)
WISCONSIN:		
Milwaukee	4,700,000 (1967)	376,000
ILLINOIS:		
Rockford	172,000 (1967)	14,000
Chicago	8,063,000 (1966)	645,000
	8,235,000	659,000
INDIANA:		
Gary	3,739,000 (1967)	299,000
Fort Wayne	98,000 (1967)	8,000
South Bend	649, 000 (1967)	52,000
Peru	54,000 (1967)	4,000
	4,540,000	363,000
MICHIGAN:		
Muskegon	2,414,000 (1967)	193,000
Saginaw	2,842,000 (1967)	228,000
Grand Rapids	75,000 (1967)	6,000
Lansing	528,000 (1967)	42,000
Kalamazoo	325,000 (1967)	26,000
Detroit	10,272,000 (1967)	822,000
Boyne City	66,000 (1966)	5,000
	16,522,000	1,322,000
OHIO:		
Cleveland	5,136,000 (1967)	411,000
Lorain	304,000 (1967)	24,000
Akron	231,000 (1967)	19,000
Youngstown	571, 000 (1967)	46,000
Toledo	2,595,000 (1967)	208,000
	8,837,000	708,000
PENNSYLVANIA:		
Erie	284,000 (1967)	23,000
NEW YORK:		
Buffalo	3,002,000 (1966)	240,000
TOTAL: GREAT LAKES	46,120,000 tons/yr	3,691,000 tons/yr

(1) Source: Keystone Coal Buyers Manual, McGraw Hill, 1967.
(2) Using 8% of coal by weight

TABLE 3.3.2

ATLANTIC COAST STUDY AREA

CURRENT POWER PLANT COAL BURN AND ESTIMATED FLY ASH PRODUCTION (1)

	Coal Burn (tons/yr)	Fly Ash Production (tons/yr) (2)
MASSACHUSETTS:		
Worcester	55,000 (1967)	4,000
Boston Area	829,000 (1967)	66,000
Fall River-New Bedford	1,357,000 (1967)	109,000
	2,241,000	179,000
RHODE ISLAND:		
Providence	217,000 (1967)	17,000
CONNECTICUT:		
	3,434,000 (1967)	276,000
OTHER AREAS:		
New York, Jersey City & Newark	10,325,000 (1967)	826,000
Bethlehem-Allentown	322,000 (1967)	26,000
Philadelphia	4,755,000 (1967)	380,000
Atlantic City	1,150,000 (1967)	92,000
Wilmington-Dover	1,278,000 (1967)	102,000
Baltimore	3,448,000 (1967)	276,000
Washington, D. C. Area	4,225,000 (1967)	338,000
Salisbury, Maryland	178,000 (1967)	14,000
Richmond Area	1,854,000 (1966)	148,000
Newport News-Hampton	518,000 (1966)	41,000
Norfolk-Portsmouth	1,967,000 (1966)	157,000
	30,020,000	2,400,000
TOTAL: ATLANTIC COAST	35,912,000 tons/yr	2,872,000 tons/yr

(1) Source: Keystone Coal Buyers Manual, McGraw Hill, 1967.

(2) Using 8% of coal by weight.

TABLE 3.3.3

1966 PRODUCTION AND UTILIZATION OF ASH

(Fly ash, bottom ash, boiler slag)

Country	Bulgaria	Denmark	United States	France	Netherlands	Poland	United Kingdom	Czechoslovakia	West Germany
Production (in 1000 tons)	900	322	25,189	3,531	366	3,270	9,256	1,100	6,200
Commercial Utilization (in 1000 tons)									
Cement (addition or replacement of hydraulic binder)	-	42.8	19.6	741	-	-	20	10	10
Cement kilns	-	-	237.7	96	-	-	-	-	-
Roads	-	37.4	184.2	601	27.8	90	1,675	50	700
Cellular concrete	20	-	-	-	-	535	347	70	50
Compacted Concrete:									
Blocks	10	25.3	151.5	78	-	195	555	55	-
Prepared concrete	-	-	146.9	-	-	-	24	-	1,050
Dams	-	-	231.3	-	-	-	-	-	-
Lightweight aggregate	-	-	259.0	-	31	-	191	-	-
Bricks	-	-	-	1	91.8	-	33	-	850
Filler on construction sites	-	62.1	483.2	187	79.8	160	1,043	50	900
Miscellaneous	20	77	1,337.4	29	1.0	10	48	-	440
Total amt. utilized	50	245.5	3,050.8	1,733	231.4	990	3,936	235	4,000
% of production used	5	76	12	49	63	30	43	21	65

Publ. by Secretariat, U. N. Eco. Comm. for Europe,
Coal Section, Energy Div., Geneva, Switzerland

An added consideration is the relatively high use factor of land in Europe leaving little available for dead storage.

The uses of fly ash in the United States are very similar to those in Europe, with additional usage here as fill for dam construction. Several experiments are under way for developing other uses, but these constitute no significant quantity at present. Progress is being made in the use of fly ash to reduce the amount of binder material in structural products. Methods for control of mine fires and subsidence prevention are advancing. Treatment of water by using fly ash as an aid to chemical coagulation and filtration is under study. A pilot study of the use of fly ash in dewatering digested sludge was successfully completed, and a demonstration plant is under construction. Experimental work here and abroad is being done on extraction of the useful minerals and trace elements from fly ash. Special attention has been given to the recovery of gallium, germanium and vanadium, but at the present time the economics of recovery do not appear favorable. Table 3. 3. 4 summarizes fly ash utilization for the United States in 1967.

The present situation on commercial use of fly ash in the U. S. is typical of an emerging industry. The limited use is primarily due to a lack of knowledge on this product by commercial enterprises. Present specifications and test methods are inadequate. Research must be performed and consolidated with other information to provide the user with practical guidelines. These problems are not insurmountable and can be solved by concerted effort. As an indication of the changing picture, the National Ash Association was formed in 1968 to promote fly ash utilization. This association provided most of the information on projected fly ash utilization in the two study regions as shown in the following table:

PROJECTED FLY ASH UTILIZATION

(Million Tons/year)

Use	Atlantic Coast Region				Great Lakes Region			
	1966-67	1975	1985	2000	1966-67	1975	1985	2000
Fill material	.06	.15	.25	.35	.08	.11	.15	.30
Light weight aggregate	.03	.20	.40	.60	.04	.15	.30	.50
Concrete additive	.14	.20	.30	.70	.19	.24	.30	.70
Other struct. products	.06	.10	.20	.40	.08	.10	.15	.40
Total ash utilized	.29	.65	1.15	2.05	.39	.60	.90	1.90

TABLE 3.3.4

FLY ASH UTILIZATION IN THE UNITIED STATES

(1967)

Use	Million Tons	Percent of Total Utilized
Concrete additive	0.600	38.5
Road & construction fill	0.300	19.2
Lightweight aggregate	0.150	9.6
Cement manufacturing	0.150	9.6
Stabilization-road base	0.120	7.7
Asphalt filler	0.120	7.7
Miscellaneous	0.120	7.7
Total utilized	1.560	100.0
Total collected	18.500	
Percent utilized	8.44	

Source: Meikle, P. G. "Fly Ash & Bottom Ash", Mining Engineering, Jan. 1969.

3.3.3 Projected Waste Quantities

From the table of power plant coal burn given in Section 3.3.1, the total amount of fly ash produced has been projected by using a uniform 8% of the coal weight. Deducting the amount for utilization, projected waste quantities are determined as shown in the following table:

PROJECTED WASTE QUANTITIES
(Million Tons/Year)

	Atlantic Coast Region				Great Lakes Region			
	1966-67	1975	1985	2000	1966-67	1975	1985	2000
Fly ash produced	2.87	3.76	4.88	3.16	3.69	4.79	6.27	4.06
Fly ash utilized	0.29	0.65	1.15	2.05	0.39	0.60	0.90	1.90
Projected waste	2.58	3.11	3.73	1.11	3.30	4.19	5.37	2.16

Examination of the difference between fly ash produced and utilized indicates that present utilization schemes will only handle a fraction of the fly ash that will be available. Hence, other uses will be sought. However, the fly ash disposal problem appears destined to increase through 1985.

3.4 WATER TREATMENT PLANT RESIDUES

Hudson (1968) has estimated that the total residues from water treatment plants is about one-tenth of the sludge that is produced in municipal waste treatment plants. While estimates of the total are of interest, a means for estimating current and future sludge production in a prescribed region is necessary for planning regional solids disposal.

Of the 100 largest cities in the United States in 1962, twenty, serving about 6 million, used ground water exclusively for water supplies. Sixty-six cities, serving 40 million, used a combination of ground and surface waters. In the Great Lakes and Atlantic Coast study areas, nealy all large cities obtained their water from surface supplies. Exceptions were Rockford, Illinois and South Bend, Indiana (Durfor and Becker, 1962).

To reduce turbidity in surface water supplies, many cities clarify and filter the water. In 1962, about 56 percent of the population in the 100 largest cities received filtered water, usually by means of rapid sand filtration. Only 28 cities serving less than 20 percent of the total population of the 100 cities employed softening processes. In the two study regions, softening was practiced only at a few cities in the Great Lakes area, namely: Ft. Wayne, Indiana; Flint, Michigan; Toledo and Youngstown, Ohio.

From the above, it is evident that residues from rapid sand filter plants utilizing surface supplies are the principal wastes emanating from water treatment plant operations in the two study regions.

3.4.1 Solids Production

The two major waste streams from rapid sand filter plants are filter backwash water and chemical coagulation residues. Characteristics of these wastes are given in Appendix A. Although the former is a large volume waste, amounting to around two percent of all the water filtered, the suspended solids concentration and biochemical oxygen demand is very low. Also as discussed in Section 4.4.2, recirculation and reuse is sometimes practiced. Hence this dilute waste is not likely to be directly included in a regional waste management system, but filter wash water solids may be included with chemical coagulation residues.

One direct method for estimating the quantity of chemical coagulation wastes produced in a specific area is to determine the quantity of solids produced per million gallons of water treated and multiply this value by the volume treated. Where water is treated for the removal of turbidity by aluminum sulfate, for example, the average turbidity and the average alum dose might be used to estimate the weight of solids-produced per million gallons. Lime is also added, but can be ignored as normally it results in the formation of soluble end products. However, it should be noted that while the coagulant dosages applied are relatively constant, the turbidity of individual surface waters may vary greatly with time.

Let us assume that an average dosage of 2.0 grains per gallon of aluminum sulfate is used in coagulation practice. This results in the precipitation of aluminum hydroxide. However, in terms of dry solids, we would produce:

$$34 \text{ mg/l} \times \frac{\text{M. W. } Al_2O_3}{\text{M. W. } Al_2(SO_4)_3 \cdot 18\ H_2O} = 34 \times \frac{102}{666} = 5 \text{ mg/l } Al_2O_3$$

If it can be further assumed that the turbidity, measured optically, can be taken as an estimate of suspended solids in mg/l, and that this turbidity is 100% removed, then an order of magnitude estimate of dry solids production per million gallons of water treated can be obtained.

In Table 3.4.1, average turbidities have been listed for a number of municipal water supplies employing chemical coagulation. Assuming the previously mentioned coagulant dosage, estimated solids production in dry lbs. per million gallons of water treated has been calculated. Water supplies drawn directly from rivers result in the highest solids production, about 600 lb/MG. Waters from lakes and reservoirs result in considerably lower solids production, about 100 lb/MG.

In the Great Lakes region over 90% of the water supply to the large cities comes directly from the lakes. On the Atlantic coast, nearly all supplies are obtained from reservoirs drawing from rivers. Exceptions include the Philadelphia plants which take directly from rivers. An average solids production of 125 lb/MG has been assumed for both regions. However, it is emphasized that actual waste production values will vary widely from plant to plant. For example, Sutherland (1969) listed sedimentation basin sludge accumulation rates in N. Y. of 45 lb/MG for Rochester, 94 lb/MG for Wolcott, and 159 lb/MG for Ithica.

TABLE 3.4.1

AVERAGE TURBIDITIES AND ESTIMATED SOLIDS PRODUCTION OF SELECTED U.S. WATER SUPPLIES EMPLOYING COAGULATION

City	Water Source	Avg. Raw Water Turbidity 1/	Precipitated Coagulant mg/l	Estimated Solids Production lbs/MG 2/
BALTIMORE, MD.	Liberty Reser.	3	5(Al)	67
PATTERSON, N. J.	Wanaque Reser.	10	5(Al)	126
ALBANY, N. Y.	Alcove Reser.	5	5(Al)	84
AKRON, OHIO	Reservoirs	5	5(Al)	84
ERIE, PA.	Lake Erie	9	5(Al)	118
BUFFALO, N. Y.	Lake Erie	12	5(Al)	143
WASHINGTON, D. C.	Potomac River	49	5(Al)	455
PHILADELPHIA, PA.				
(a) TORRESDALE	Delaware River	126	5(Al)	1100
(b) BELMONT	Schuykill River	32	5(Al)	310
CINCINNATI, OHIO	Ohio River	70	7(Fe)	650
CHICAGO, ILL.	Lake Michigan	15	5(Al)	168

1/Reference: Durfor and Becker, Public Water Supplies of the 200 Largest Cities in the United States, 1962.

2/Turbidity assumed as mg/l of suspended solids and aluminum oxide (Al_2O_3) estimated as 5 mg/l or ferric oxide (Fe_2O_3) estimated as 7 mg/l.

3.4.2 Surface Water Consumption

For 1965, the following water use data pertaining to public supplies is available (Murray, 1968):

Region	Population Served by Surface Water (millions)	Surface Water Use (MGD)
Eastern Great Lakes-St. Lawrence	9.84	1600
Western Great Lakes	8.19	1800
	18.03	3400

Per Capita Surface Water Consumption = 188 gallons/day

New England	6.81	950
Delaware-Hudson	16.80	2300
Chesapeake	5.78	840
	29.39	4090

Per Capita Surface Water Consumption = 139 gallons/day

Although the water-use regions cover somewhat different areas than the study regions outlined in Section 3.1, the per capita water consumption figures given should be indicative of the general consumption in either case. For 1968, 140 gpcd was taken for the Atlantic Coast Study region and 190 gpcd was used for the Great Lakes Study region.

For projecting future water consumption, data on overall U. S. withdrawals for public water supplies during the period 1950-1965 was examined. This data indicated a general increase in per capita consumption of about 15% over this period. The same general rate of increase, i. e., about 1% per year from 1968 through the year 2000 has been used for projecting future surface water consumption rates.

3.4.3 Projected Chemical Coagulation Wastes

In Table 3.4.2, total populations have been listed for each study region in the years 1968, 1975, and 2000. These populations were derived previously in Section 3.1. Assuming the percentage served by surface water supplies is constant at 90% in both regions, the population served by surface water supplies is determined. Projected per capita surface water use was then established for 1975 and 2000, as discussed above.

With populations served and per capita water rates established, the total surface water use was determined. A derating factor was then applied to account for the surface water supply that is not clarified. In the Great Lakes, clarification is widely practiced and so a 90% factor was taken. In the Atlantic Coast region, New York, Boston and a number of smaller communities omit clarification for all or part of their water supply and a 60% factor was used. Finally, assuming the constant 125 lbs. solids production per million gallons, the amount of chemical coagulation residues, in tons per day of dry solids, was calculated.

TABLE 3.4.2

Projected Waste Quantities

Chemical Coagulation Residues

	1968	1975	2000
Great Lakes Study Region			
Total Population (millions)	22.55	23.80	30.89
% served by surface water	90	90	90
Population served by surface water (millions)	20.3	21.4	27.8
Per capita surface water use (gpcd)	190	205	260
Total surface water (M.G.D.)	3860	4390	7230
% surface water clarified	90	90	90
Total surface water clarified (M.G.D.)	3470	3950	6500
lbs. per M.G.	125	125	125
Tons of dry solids per day	217	247	407
Tons of dry solids per year	79,000	90,000	149,000
Atlantic Coast Study Region			
Total Population (millions)	36.41	38.84	50.17
% served by surface water	90	90	90
Population served by surface water (millions)	32.8	35.0	45.2
Per capita surface water use (gpcd)	140	150	190
Total surface water (M.G.D.)	4600	5250	8600
% surface water clarified	60	60	60
Total surface water clarified (M.G.D.)	2760	3150	5160
lbs. per M.G.	125	125	125
Tons of dry solids per day	173	197	323
Tons of dry solids per year	63,000	72,000	118,000

4.0 PRESENT METHODS AND COSTS OF DISPOSAL

4.1 DIGESTED SLUDGE

For ultimate disposal, wastewater sludges must be discharged to one of the three sectors of the environment---the land, the water, or the air. In this section, current methods of disposal are reviewed and typical disposal costs are presented. Land utilization and ocean disposal practices are covered only cursorily because of their more extensive treatment elsewhere in this report.

4.1.1 Disposal to the Land

Techniques for ultimate disposal of sludge to land include stockpiling of residues and beneficial application of sludges to agricultural lands. Examples of use of land for stockpiling include disposal of liquid or dewatered sludge to permanent lagoons, landfills, mines and quarries. Raw primary, activated sludge or digested sludge can be disposed of in this manner under strict control, but usually the sludge is digested to minimize environmental problems.

Lagoon disposal of digested sludge has been widely practiced. The operation is simple, flexible and suitable for plants of varying size. However, few large metropolitan plants presently use this method because large available land areas, relatively close to the treatment plant, are practically nonexistant.

The other type of land disposal involves use of sludge agriculturally for conditioning of soil and fertilization of plants. The intent of these practices is not to compete with commercial fertilizations, but to accomplish economical waste disposal in a beneficial manner. Sludge is applied to land in three forms---liquid, dewatered or dried.

Because of the difficulty in dewatering most waste sludges, disposal of sludge in the liquid form is often advantageous. To avoid odor and nuisance problems, the decomposible organic content is normally reduced by anaerobic digestion prior to land application. A major benefit of this method of disposal is that nutrient removal in the treatment plant is improved because liquid removed from the sludge is not recycled. Also, the liquid serves as a source of irrigation water.

Sludge dewatered by vacuum filtration or, less frequently, by centrifugation or other methods, also may be spread on land. Raw primary sludge or raw primary and secondary sludge is sometimes distributed, but again the sludge is usually digested before it is dewatered. The solids concentration of dewatered digested sludge is typically about 25 to 30 percent by weight.

Drier sludge, from drying beds or lagoons, may also be spread on land. The practice is common at small installations where the air dried sludge is often stockpiled and made available for soil conditioning and fertilizing.

Of the various forms in which sludge has been applied to land as a means of disposal, the heat dried form has the best record in terms of public acceptance. Rotary or flash driers are used for drying with dewatering normally preceding. The moisture content of sludge is reduced to about ten percent in the drying process, but combustion of organics does not occur. Heat drying serves to remove the hazard of disease transmission (Connell and Garrett, 1963) and renders the sludge suitable for packaging, shipping, and storing.

Waste activated sludge often has been used in heat drying installations because its fertilizing value exceeds that of other types of sludges. Also, application of heat drying systems has normally been limited to larger cities where the extensive processing and marketing required for commercial sale of fertilizers could be justfied. Chicago, Milwaukee, and Houston are examples of cities which heat dry sludges and all three dry waste activated sludge.

4.1.2 Disposal to Water

The discharge of sludges to inland waters is not a satisfactory means of disposal under present water quality standards and anti-degradation laws. However, ocean disposal is being used by a number of coastal cities in the U.S.A. Currently, two methods are in use, namely, disposal by surface vessels (barges and tankers) and disposal by ocean outfall pipelines. The practice is usually associated with the disposal of digested sludge. However, a number of cities in Florida and other states are still discharging raw sewage to the ocean.

Ocean disposal operations employing surface vessels typically involve sludge storage, followed by periodic transport to the disposal site. In order to reduce transportation costs, some plants concentrate the sludge prior to disposal. Philadelphia, Camden, Elizabeth, Yonkers and Newark all barge sludge to sea. At Philadelphia's Northeast Plant, digested sludge is hydraulically dredged from lagoons at a concentration close to 10 percent solids by weight. The material is pumped several days a week to a 2700 ton seagoing barge which is towed, under contract, 110 miles down the Delaware River to a disposal site some 10 miles out in the Atlantic. The depth at the dumping site is approximately 60 feet.

New York City utilizes a fleet of six self-propelled sludge tankers, the largest of which carries a load of over 3300 tons. Eleven treatment plants are served with an average round trip of about 60 miles. The sludge averages about 5 percent solids by weight. It is currently digested at all plants except Wards Island and this plant will soon conform to the others. Disposal is in water about 10 miles from the nearest land and more than 70 feet in depth.

Coastal water pollution control plants which employ outfall lines to dispose of sludge have two principal alternatives. They may use a separate outfall for sludge, or they may discharge the sludge periodically with the treated wastewater through a single outfall line.

At the City of Los Angeles Hyperion Plant, digested sludge is discharged through a separate 7 mile, 22" concrete lined steel pipe. It is presently diluted with plant effluent to a solids concentration of less than one percent by weight with the expectation of increasing this concentration as the quantity of digested sludge and supernatant increases in the future. The outfall line terminates at a depth of 320 feet.

Among the areas discharging sludge with effluent are Boston and the L. A. County plants. Increased dispersion and easier handling of sludge are benefits claimed for this alternative. Often the sludge discharge is done intermittently to take advantage of strong currents, ebbing tides or non-bathing periods.

In reviewing present ocean disposal operations, it is concluded that there is little evidence to date to support any claim that sludge disposal at sea is degrading the ocean environment. However, there is also a lack of detailed monitoring at most disposal sites. Furthermore, long term effects are completely unknown. The Hyperion plant in Los Angeles has conducted the most thorough water quality monitoring program, but the sludge outfall has only been in operation for 12 years.

4.1.3 Disposal to the Air

Disposal of sludges to the air is accomplished by combustion processes. These processes are "ultimate disposal" techniques only for combustible materials. Since about 30 percent of sludge solids are nonvolatile, considerable quantities of ash remain as residues. Ultimate disposal of this ash can present an appreciable problem (Walters and Ettelt, 1965). In addition, the discharge of some of the combustion products to the atmosphere may create air pollution problems.

Because combustion processes, in general, render sludges inoccuous, pre-treatment to alter sludge characteristics is not necessary. At some plants, anaerobic digestion has been provided prior to combustion of solids with methane gas from the digestion process being used as fuel in the combustion process. Burd (1968) has concluded that this approach is not economically justified.

Pre-treatment to increase the concentration of solids prior to their combustion is, however, very important, as efficient performance requires that the process be thermally self-sustaining. Most dry combustion processes require sludge concentrations in the range of those achieved by sludge dewatering techniques such as vacuum filtration. To prepare the sludge for dewatering, elutriation is commonly practiced. The resulting elutriate and filtrate often contain a high concentration of solids and their recycling can create a major additional load on the treatment plant.

Reviews of available types of sludge incineration equipment have been presented recently by Frankel (1966), Robertson and Woodruff (1967) and Burd (1968). The type most frequently used in waste treatment plants is the multiple hearth incinerator. Other combustion techniques, such as fluidized beds, atomized suspension techniques, and the wet oxidation process, are also available, but they are not in widespread use.

4.1.4 Disposal Costs

The cost of sludge disposal from a given area is dependent on the type of sludge, the method of disposal, the population served, and the distance to the ultimate disposal site. In a general examination of the comparative economics of sludge disposal, such as this one, a description of the three dimensional response surface with size, distance, and cost as variables are needed for each type of disposal. The problem is further complicated by the fact that various methods of conditioning, concentrating, or dewatering could be considered for each disposal scheme. Several general analyses of sludge disposal costs have appeared in recent years, and these have been used for the most part in the paragraphs which follow, in lieu of descriptions of sludge disposal costs at individual installations. However, it must be emphasized that there is a lack of uniform accounting and reporting procedures. Therefore, disposal costs which appear in the literature can only serve as a general guide.

The relative cost of various methods of sludge disposal for cities of various size as a function of distance to the point of disposal has been evaluated by Riddell and Cormack (1966). Some of their results are shown in Figures 4.1.1, 4.1.2 and 4.1.3. All costs are on the basis of dollars per ton of suspended solids removed in the treatment process.

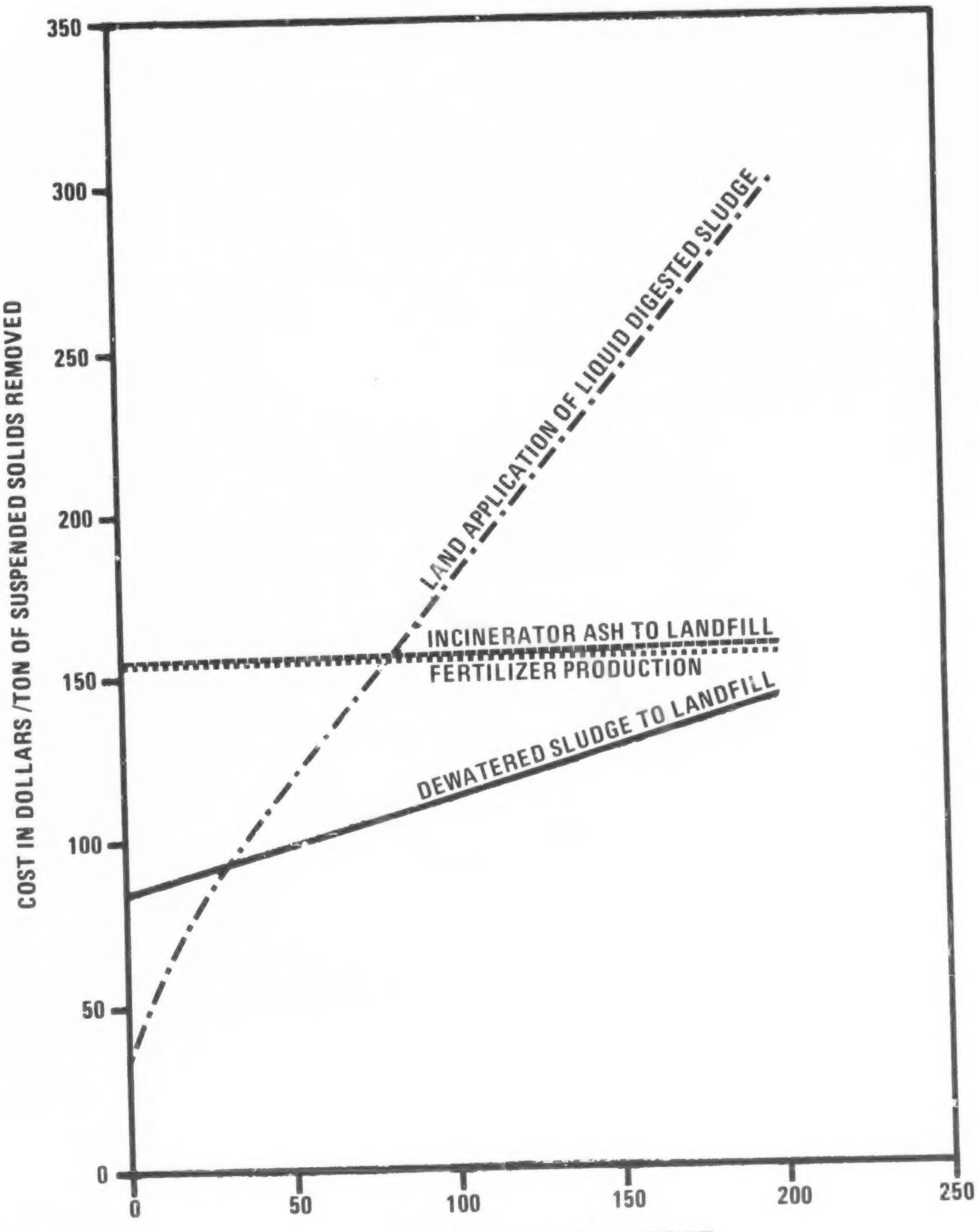
Thickening and digestion were included in all schemes. Disposal by land application of liquid digested sludge assumed spreading a 3.5% sludge from tank trucks with transportation by tank truck for a city of 10,000 and pipeline for larger installations. Dewatered sludge at 25% solids and incinerator ash were considered to be transported by truck. Fertilizer was considered to be produced by heat drying digested sludge, and its value was taken as $5 per ton.

Riddell and Cormack's data suggest that liquid spreading of digested sludge is the most economical means of sludge disposal for a city of 100,000 if a disposal site is available within 30 miles. This distance is extended to about 100 miles for a city of 1,000,000. Incineration and fertilizer production are feasible only for large installations when disposal sites are more than 100 miles away. Land disposal of dewatered sludge is the most economical means of disposal for communities up to 100,000 people for disposal sites over 30 miles and less than 200 miles away.

Experience with a variety of sludge disposal methods at the Metropolitan Sanitary District of Greater Chicago gives a comparison of costs for

FIGURE 4.1.1

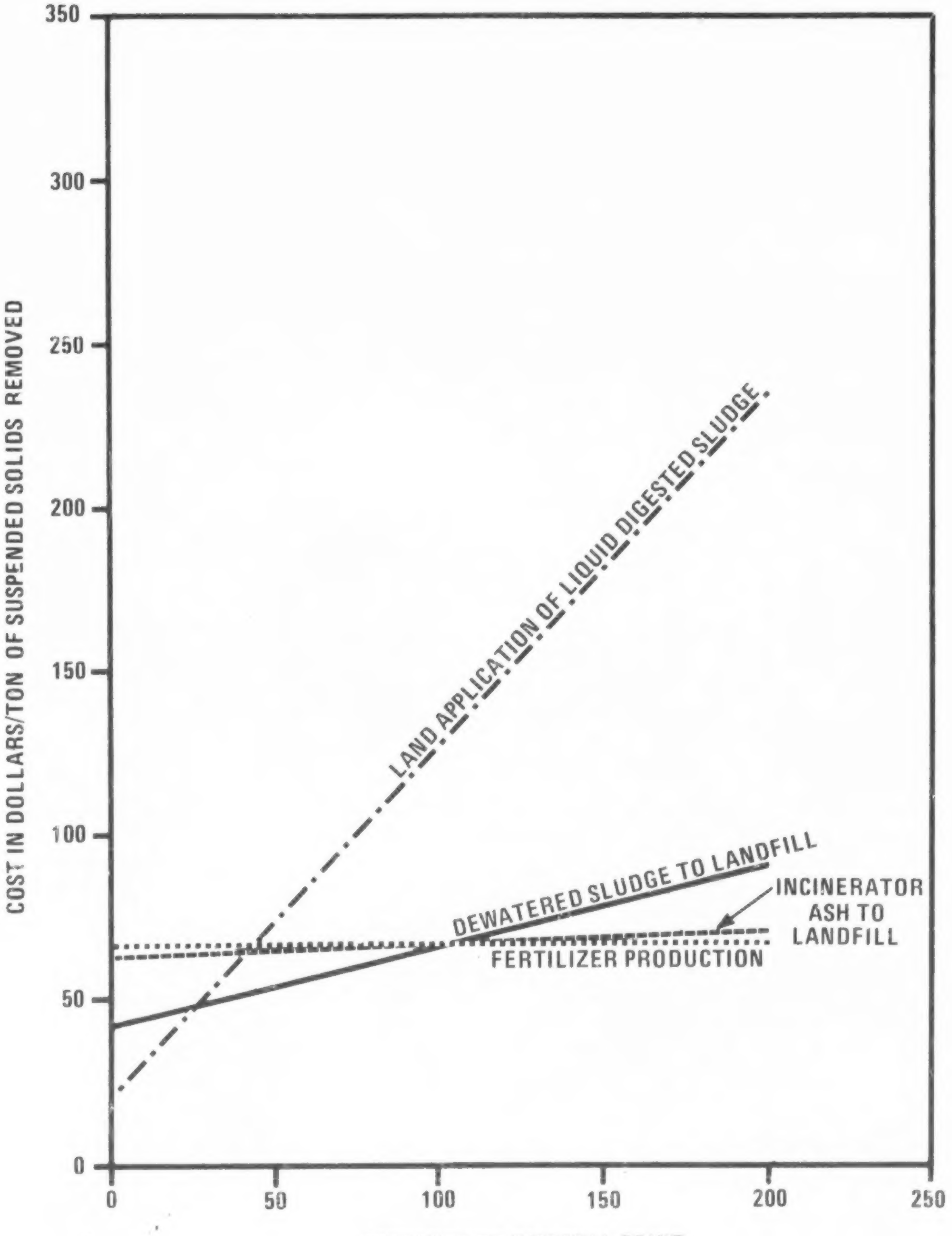
COST OF SLUDGE DISPOSAL BY VARIOUS METHODS
FOR CITY OF 10,000 PEOPLE



(After Riddell and Cormack, 1966).

FIGURE 4.1.2

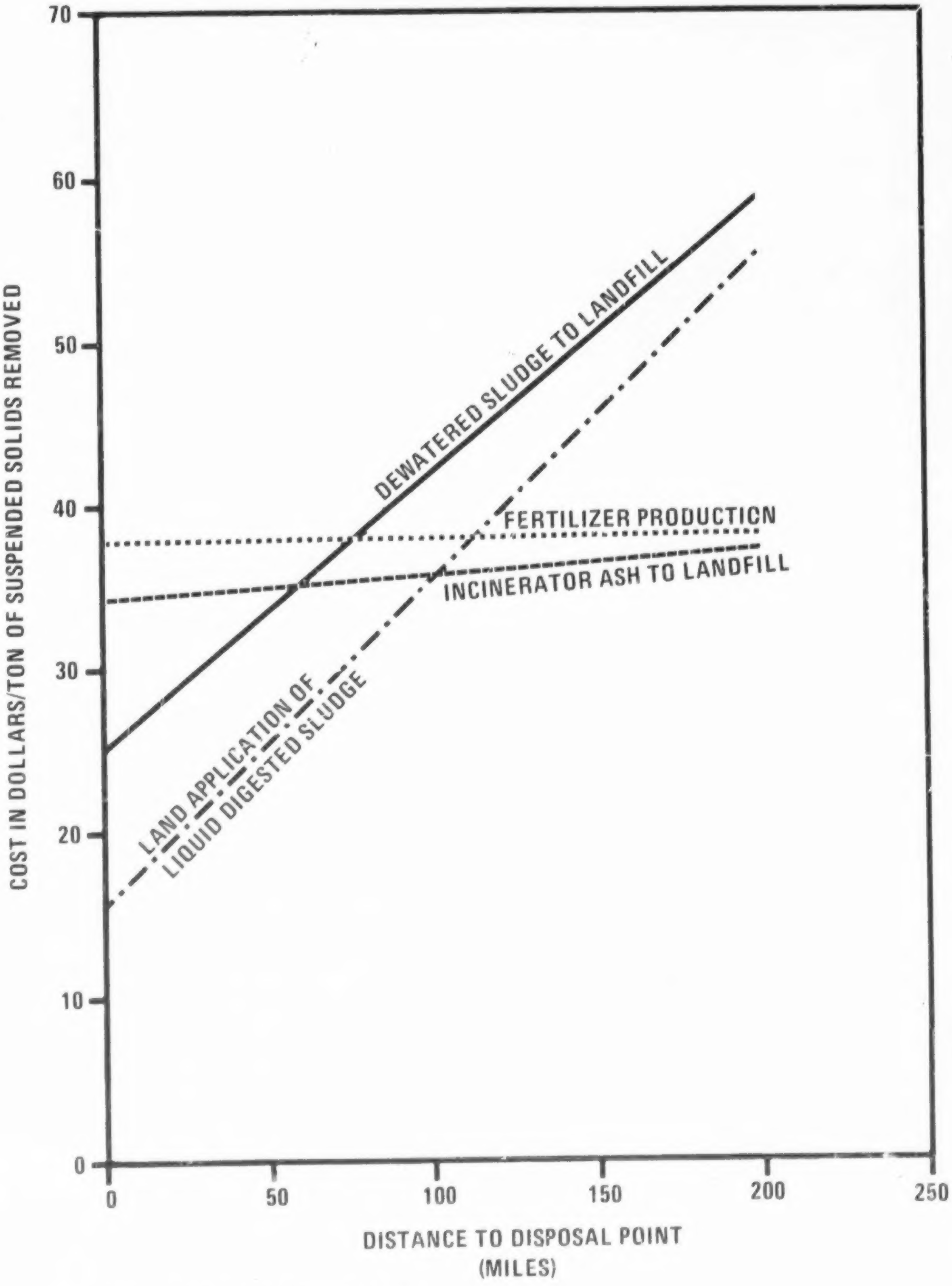
COST OF SLUDGE DISPOSAL BY VARIOUS METHODS
FOR CITY OF 100,000 PEOPLE



(After Riddell and Cormack, 1966).

FIGURE 4.1.3

COST OF SLUDGE DISPOSAL BY VARIOUS METHODS FOR CITY OF 1,000,000 PEOPLE



(After Riddell and Cormack, 1966).

alternate schemes. Bacon and Dalton (1968) have presented the following summary of costs:

Method	Disposal Cost $/dry ton of digested sludge
Wet Air Oxidation of Digested Sludge	50
Digestion, Dewatering, and Incineration	57
Application of Liguid Digested Sludge on Agricultural Land	15
Digestion and Permanent Lagoons	49
Strip Mine Reclamation with Liquid Digested Sludge	16
Drying and Sale as Fertilizer (net cost on equiv. basis)	45

Costs for strip mine reclamation and for agricultural use of liquid digested sludge were estimated for disposal sites about 50 miles away and other costs were based on actual operating records. Digestion costs comprise $12.50/ton of the totals shown.

Recent operating costs of ocean disposal by tanker at New York are reported to be about $13.00/ton. With a much higher sludge concentration and only two pickup points, Philadelphia's contract barging costs are close to $9.00 per ton. In comparison, Hyperion sludge outfall costs are estimated at around $4.50 per ton of dry solids, of which approximately one-fourth is operating expense. Digestion costs are excluded in all cases. For ocean disposal, the distance to the disposal site is obviously a major factor. At present, all sludge disposal in the Atlantic is relatively near to shore, around 10 miles from land. If requirements dictated removal to the edge of the continental shelf, some 80 miles from shore, a large escalation in costs could be expected.

An exhaustive summary of reported sludge treatment and disposal costs has been prepared by Burd (1968). His summary of the cost of individual sludge handling processes is given in Table 4.1.1. Burd also presented a tabulation of the overall cost of various methods of ultimate disposal. These costs include the cost of pre-treatment such as digestion, dewatering, etc., and are shown in Table 4.1.2.

In conclusion, sludge disposal costs vary widely. Comparative economic studies from the literature serve only as a guide. A thorough economic evaluation of each alternative for a particular treatment plant is required to obtain a true cost picture. However, numerous published reports have indicated that disposal to the land or ocean is often the cheapest.

TABLE 4.1.1

UNIT SLUDGE PROCESS COSTS

(after Burd, 1968)

Process	Capital and Operating Costs ($/dry ton) Average	Range
Thickening		
gravity	-	1.50 - 5
flotation (1)	-	6 - 15
centrifugation (1)	-	3 - 20
Dewatering		
vacuum filtration	15	8 - 50
centrifugation	12	5 - 35
sand drying beds	-	3 - 20
Anaerobic Digestion	-	4 - 18
Elutriation	-	2 - 5
Lagooning	2	1 - 5
Landfilling (2)	-	1 - 5
Pipeline Transportation (3)	5	-
Liquid Disposal on Land	10	4 - 30
Heat Drying	35	25 - 40
Incineration	20	8 - 40
Barging to Sea	10	4 - 25

(1) Wide variation depending on conditioning chemical requirement.

(2) Long hauls higher.

(3) Moderate distance.

TABLE 4.1.2

TOTAL SLUDGE DISPOSAL COSTS

(after Burd, 1968)

Method of Ultimate Disposal	Capital and Operating Costs ($/dry ton)	
	Average	Range
Incineration		
Wet Oxidation	42	-
Multiple Hearth	30	10 - 50
Fluidized Bed	30	10 - 50
Landfill (dewatered sludge)	25	10 - 50
Lagooning	12	6 - 25
Land Application (1)		
Heat Dried Sludge	50	40 - 55
Dewatered Sludge	25	10 - 50
Liquid Sludge	15	8 - 50
Disposal at Sea		
Barging	12	5 - 25
Pipeline	11	-

(1) Excludes any return from sale of sludge

4.2 MAINTENANCE DREDGINGS

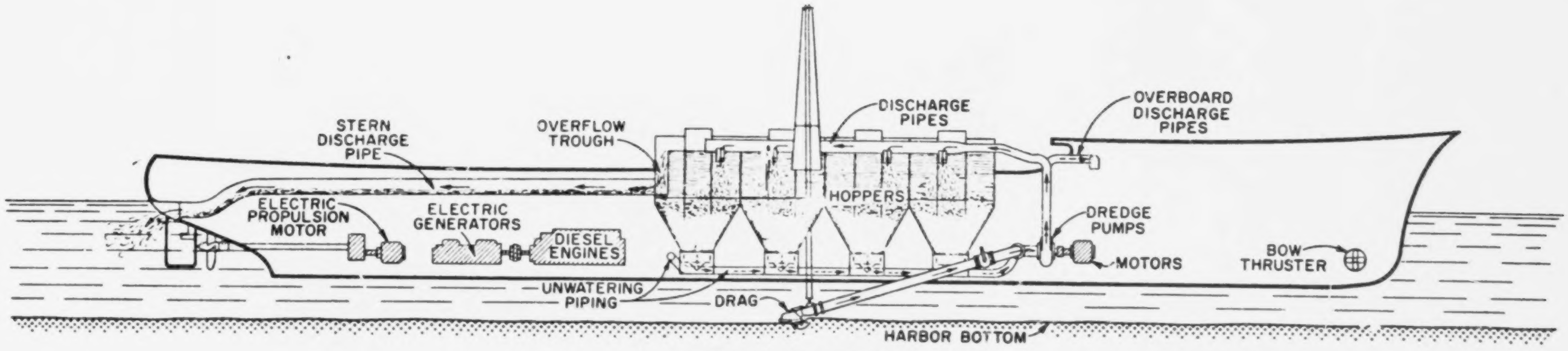
The dredging operation involves three distinct phases, namely excavation, transportation and disposal. Various types of dredging equipment are used but they can be classed as either mechanical or hydraulic. When mechanical equipment is used, auxiliary equipment consisting of scows and tugs are required for the transportation-disposal process. Examples of this class are dipper and clamshell dredges.

With hydraulic equipment, excavation, transportation and disposal are all performed with the same piece of equipment. The hopper dredge is the prime example of this group. Mauriello and Caccese presented an excellent paper in 1963 on hopper dredge disposal techniques. They stated that approximately 65 million cubic yards of material (measured in place) were removed annually by these dredges in the various coastal and Great Lakes regions of the United States. For the Great Lakes alone, hopper dredges presently do about 75 percent of the approximate ten million cubic yards of annual maintenance dredging (U. S. Army Corps of Engineers, 1969). Figure 4. 2. 1 is an operational sketch of a typical hopper dredge.

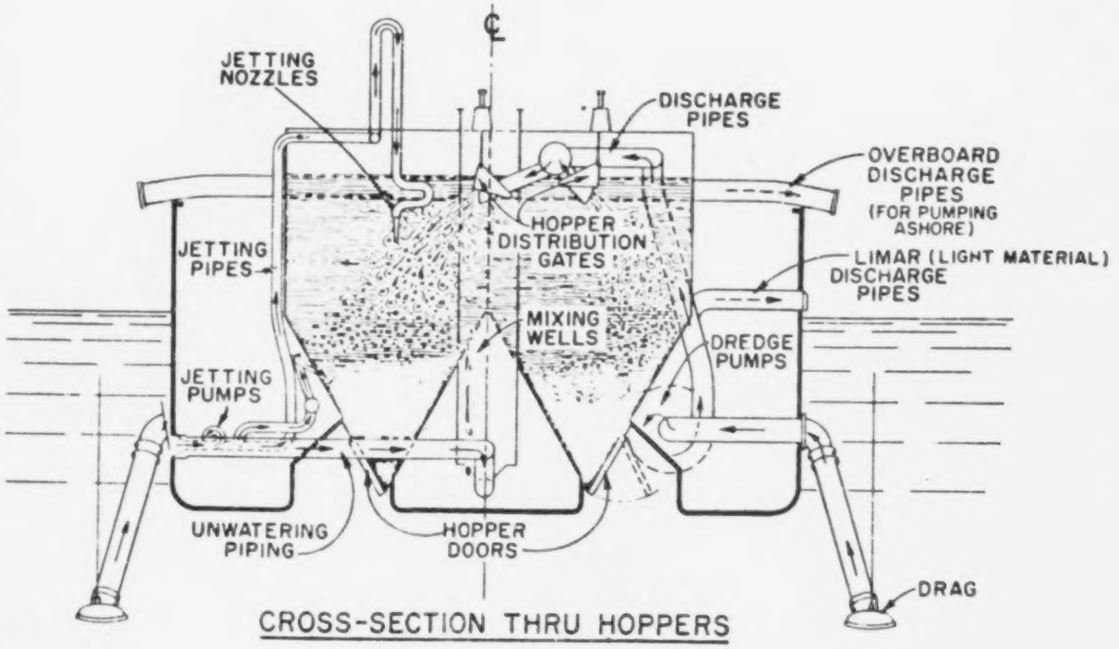
4.2.1 Removal to Deep Water

Removal of maintenance dredgings to deep water outside of the navigation areas has traditionally been the chief method of disposal. Dipper, clamshell or hopper dredges are used for removal of the sediments. In general, the dumping areas are selected as close as possible to the work site with avoidance of water supply intakes or recreational beaches. However, the effects of disposal in the receiving area are largely unknown.

Dipper dredges are often utilized for harder material, while clamshell dredges are usually used for excavation of soft or cohesive materials. The latter are particularly useful where sizeable depths are encountered or when the excavation must be performed in close quarters alongside structures. Winding river channels are usually handled with a clamshell dredge. Rates of dredging as high as 400 cubic yards per hour have been obtained with both dipper and clamshell dredges. In the Great Lakes, about 20 percent of the annual maintenance work is accomplished by clamshell dredges.


STERN DISCHARGE PIPE
OVERFLOW TROUGH
DISCHARGE PIPES
OVERBOARD DISCHARGE PIPES
ELECTRIC PROPULSION MOTOR
ELECTRIC GENERATORS
DIESEL ENGINES
HOPPERS
DREDGE PUMPS
MOTORS
BOW THRUSTER
UNWATERING PIPING
DRAG
HARBOR BOTTOM

LENGTHWISE SECTION


℄
JETTING NOZZLES
DISCHARGE PIPES
OVERBOARD DISCHARGE PIPES (FOR PUMPING ASHORE)
JETTING PIPES
HOPPER DISTRIBUTION GATES
LIMAR (LIGHT MATERIAL) DISCHARGE PIPES
MIXING WELLS
DREDGE PUMPS
JETTING PUMPS
UNWATERING PIPING
HOPPER DOORS
DRAG

CROSS-SECTION THRU HOPPERS

SYMMETRICAL ABOUT ℄

OPERATIONAL SKETCH OF HOPPER DREDGE

Figure 4. 2. 1

From U. S. Army
Corps of Engineers
Buffalo District

Hopper dredges remove the bulk of maintenance dredging materials. In this operation, suction pipes are lowered from the ship's side and the drag head at the pipe's intake slides over the sediments to be dredged. In some materials, air or water is injected in the suction area to assist the breaking up and suspending of the sediment. However, this equipment is particularly efficient in removing a thin layer of sediment covering an extensive area. They are often employed when wave action or heavy traffic in channels and harbors makes a stationary plant unsuitable or when the disposal area is far removed from the dredging site.

In the usual operation, the sediments are pumped as a water slurry directly into hoppers where the material settles to the bottom. Hopper capacities range from about 400 c. y. to over 8,000 c. y. Pumping continues beyond the point of filling the tanks in order to increase the amount of captured solids. The excess water, along with some suspended material, is discharged overboard. In the case of heavy materials, relatively minor quantities of solids are lost in the overflow from hopper dredges. However, with materials such as fine silt, the percentage of solids lost by pumping after the hoppers are filled with the dredged mixture may be significant. Generally, material balances are not performed. However, one series of tests in 1967, while the hopper dredge "Hoffman" was operating on the Rouge River, indicated that about 24 percent of the total intake solids was lost as overflow (Unpublished report, Corps of Engineers, 1968). It must be emphasized that this figure should not be considered representative, as the overflow would be expected to vary from harbor to harbor, day to day, and even load to load.

One additional method for removal of sediments is by agitation dredging. In this process the dredge intentionally discharges the material overboard. The objective is to transport and deposit the sediment outside of the channel limits by tidal, river or littoral currents. This method can be very attractive for interim dredging improvements, provided that the composition and characteristics of the sediments along with the prevailing currents are such that the material will not deposit in the immediate vicinity of work. However, since nothing is permanently removed, the long range effectiveness of this method requires careful study.

4.2.2 Disposal to Landfills and Offshore Dike Areas

One method for land disposal of maintenance dredgings is direct pumpout from hopper dredges or the utilization of a pipeline dredge. Where deep water disposal sites are relatively far, this method can be an economical alternative. In the Great Lakes area, maintenance dredgings from the Rouge River at Detroit and the Maumee River in Toledo have been deposited in diked disposal areas since 1960 and 1961, respectively.

The greatest problem in utilizing this procedure is acquiring the nearby areas for disposal. Often the sediment is an unsatisfactory fill as it is hygroscopic and lacks shear strength. Other limiting factors are the interference of the discharge line with navigation and the distance which the dredge pumps can move the material. One-half mile discharge lines are presently the effective limit of small pipeline dredges, although large pipeline dredges or direct pumpout hopper dredges can economically pump as much as four miles (Cable, 1969).

A second procedure with hopper dredges for land disposal is the sump rehandling technique. Here the hopper dredge unloads the sediments through a self-contained and elevated discharge line into a rehandling barge or vessel. The rehandling vessel then disposes of the material by pumping it behind shore disposal area dikes. For maximum economy, sometimes two hopper dredges will work in conjunction with one sump rehandler vessel. Although this operation can dredge mass quantities of material efficiently, few projects are large enough to justify the mobilization and operating cost of this dredging plant. The major application of the sump rehandling technique has been the principal waterways of Philadelphia and Norfolk (Mauriello and Caccesse, 1963).

4.2.3 Disposal Costs

As can be expected, disposal costs are highly variable and cannot be simply classified by the various types of dredges. The material characteristics, the site of work, and the method of disposal are all important factors. Hauling distances may range widely. For example, the average hauling distance for Corps of Engineers hopper dredging averages a few miles, but all spoil dredged in the New York Harbor is dumped in a designated area in the Atlantic which involves a haul exceeding 30 miles from some locations (Mauriello and Caccesse, 1963).

Data on all Corp-wide hopper dredging during the period fiscal years 1954 through 1966 indicates an average cost of around \$0.25 per cubic yard (bin measurement) for an average haul distance of slightly less than 3.5 miles (U.S. Army Corps of Engineers, 1967). In comparison, Great Lakes costs for dipper dredges, which are employed on the hardest types of materials, range upwards from \$2.00 per cubic yard. Pipeline dredges usually remove material for under \$1.00 a cubic yard, while clamshell dredging costs range between \$1.00 and \$2.00 per cubic yard (U.S. Army Corps of Engineers, 1969).

4.3 FLY ASH

4.3.1 Disposal Methods

The tendency of power companies to build mine-mouth plants somewhat diminishes the fly ash disposal problem in the Great Lakes region. However, older power plants in the metropolitan areas still constitute a significant source of fly ash waste for which disposal facilities must continue to be provided. A somewhat different situation exists in the Atlantic Coast region, as the trend to mine-mouth generation does not exist. Also, the problem is accentuated by the increased population densities and land use.

Lagooning was once the principal fly ash disposal method. However, pollution problems would arise when leaks occurred in the dikes or when the lagoon overflowed. The method commonly used at present is trucking to nearby borrow pits or other low-lying areas where the surface elevation can be raised. By this method, the company hopes to offset fly ash disposal costs by appreciation of the fill land. As the population density increases and land use becomes more intense, this alternative will no longer be feasible. A few plants along the Atlantic Coast presently dispose of their ash material at sea.

In-plant transportation of fly ash is usually by hydraulic or pneumatic piping systems. Fly ash transported pneumatically is collected in a dry state electrostatically or mechanically. It is stored in silos and then loaded into trucks or rail cars. On-site storage capacity is always provided because of the possibility of interruption of transportation to the ultimate receiving site. If hauling is by open vehicle, water must be added to prevent top losses during transit.

Bottom ash is nearly always collected in a sump from which it is pumped to a lagoon. There the water is decanted and ash removal to final disposal is generally mechanical. Present trends in the power industry indicate that wet ash boilers will be replaced with completely dry fireboxes, producing only dry fly ash.

Both hydraulic and pneumatic transport methods discussed above have been in service for many years at a number of plants. All piping systems are relatively short, generally a few hundred feet in length. Therefore none would fall into the category of long distance pipelines.

4.3.2 Disposal Costs

Table 4.3.1 lists fly ash land disposal costs for plants in various selected areas based on a recent study. This cost is an approximation of handling expense and does not reflect the value of the receiving land. For disposal to sea, costs range in the area of $1.50 to $2.00 per ton.

TABLE 4.3.1

ESTIMATED 1967 FLY ASH LAND DISPOSAL COSTS

Area	Cost ($/Ton)*
New England	$ .60 - .80
New Jersey	.50 - .60
New York City	1.10 - 2.50
Western New York	.60 - .70
Philadelphia	.60 - .70
Cleveland	.70 - .80
Detroit	.70 - .80
Northern Indiana	1.30 - 1.40
Wisconsin	1.30 - 1.40
Baltimore	1.00 - 1.20
Ohio Valley	.40 - .50
Dayton	.70 - .80
Bethlehem - Allentown	.30 - .40
U. S. average for disposal	$ .50

*Handling Costs Only

Source: Electrical World, August 21, 1967, pg. 61 - 63.

4.4 WATER TREATMENT PLANT RESIDUES

As discussed in Section 3.4, the major water treatment plant residues originating from large cities are those associated with the operation of rapid sand filter plants. Chemical coagulation residues were singled out as the most significant waste produced, while filter backwash water was mentioned as being of less concern. In this section a brief survey of current disposal methods and costs is presented for both waste products.

4.4.1 Chemical Coagulation Wastes

A questionnaire sent in 1952 by the American Water Works Association to sanitary engineers throughout the United States, yielded the following data on the disposal of residues from 1530 water treatment plants excluding those plants which practice primarily water softening (Dean, 1953):

Disposal of Chemical Coagulation Residues

Method of Disposal	Percent
To streams or lakes	92.4
To storm sewers or surface drains	3.5
To sanitary sewers	0.3
To sludge beds	3.1
To city reservoirs, irrigation ditches, dry creeks, impounding basins	0.8

It is apparent that the discharge of water treatment plant residues to surface waters was almost a universal (96%) practice at that time. No evidence has appeared since to indicate that the situation has radically changed. However, with the advent of more stringent water pollution control legislation, this practice seems destined to cease shortly. Also, with the trend to higher filter and basin loadings, continuous sludge removal is being accelerated, which increases the feasibility of discharging the coagulation residues to sanitary sewers.

Other data of interest obtained from the questionnaire pertain to the arrangement of filter plants and methods of removing residues from settling basins (Dean, 1953). Three-quarters of the filter plants surveyed employed coagulation and sedimentation prior to rapid sand filtration; 17% employed filtration alone, generally by slow sand filters; 6% employed presedimentation prior to coagulation and sedimentation. As for solids removal, 91% of the plants employed flushing and fire streams for intermittent solids removal, the other 9% of the settling basins had mechanisms for continuous solids removal. In addition, the respondents to the questionnaire indicated that there was no demand for the waste solids as a soil conditioner.

4.4.2 Filter Backwash Water

The 1952 survey of disposal techniques at 1,699 water treatment plants indicated, as shown below, that filter washings were disposed of in the same general fashion as residues from chemical coagulation (Dean, 1953):

Disposal of Filter Washings

Method of Disposal	Percent
To streams or lakes	82.6
To storm sewers or surface drains	10.7
To sanitary sewers	2.5
To lagoons from which solids are not removed	1.9
To dry creeks	0.8

Since filter washings normally have less than 0.10 % suspended solids, there was little objection to discharging them into streams or lakes in the past. Moreover, the fact that a number of water filtration plants chlorinate prior to filtration, usually results in a higher bacterial quality in the washwater than in the raw water. However, in recent years, activated carbon use has grown. As a result, complaints on direct discharge of filter washwater to the stream have increased.

Numerous plants now return the filter washwater to the raw water at the mixing basins. In some cases, the filter washwater is partially clarified before return to the head end of the plant. An example of this practice of recirculation and reuse is Springwell, Detroit's largest municipal water treatment plant (Remus, 1966). This plant has a capacity near one-half billion gallons of water per day. It is operated so that all of its washwater, approximately 2 percent of its total flow, is drawn into a 0.5 million gallon reservoir and then recirculated into the rapid mixing basins for reuse. The accumulated solids are removed from the settling basins twice annually and are discharged to the municipal sewers.

During a 3-month period of low turbidities, the practice of recirculation resulted in filter runs averaging 36 percent longer in the plant when backwash water was recycled. Final turbidities and chlorine demand were unaffected. During periods of high turbidities, there were no differences in the results, with or without recycling.

The important advantage of recycling was the saving of approximately $40,000 annual sewer charges plus an estimated $7,000 in pumping costs due to longer filter runs. Disadvantages of recycling included the build-up of plankton in the settling basin, the increased accumulation of solids in the settling basin, and the required coordination of filter backwashing and recycling.

5.0 ENVIRONMENTAL CONSIDERATIONS FOR REGIONAL LAND DISPOSAL SYSTEMS

Ultimate disposal of wastes must be accomplished in a manner which minimizes offense to the environment and to the aesthetic senses. Considered in this section are the possible effects of regional waste management systems on the land environment. Emphasis is placed on digested sludge as disposal of this waste is considered to be the prime reason for establishing a regional system. Also, sludge disposal presents most of the problems that will occur with other waste products. Disposal of fly ash and water treatment plant coagulation residues present simpler problems because of the nature of these wastes and the far smaller volume that is produced. Although there is little experience with land disposal of dredgings, environmental considerations for disposal of sludge may be considered applicable to maintenance dredgings as well.

5.1 EFFECT ON WATER ENVIRONMENT

Application of waste sludge to land can influence the quality of both groundwater and surface water. However, the possibility of water quality impairment because of land usage is not unique to areas which receive waste sludges. Modern agricultural practices also create opportunities for introduction of sediment, plant nutrients, pesticides and other materials into the water environment (Wadleigh, 1968).

The potential effect of waste sludges on the water environment is highly dependent on local conditions. Soil types, moisture conditions, type of vegetation, climatic conditions, sludge distribution practices, and cultivation techniques all influence the response of a particular soil system to sludge application. Furthermore, evaluation of the potential of sludge constituents to influence water quality, particularly groundwater quality, is complicated by changes in the nature of the sludge constituents in the soil. In this section, the fate of several of the more important biological and chemical constituents of waste sludges in soils is considered.

5.1.1 Pathogenic Microorganisms

Although digested sludge cannot be considered free of microbial disease agents, the anaerobic digestion process produces a significant reduction in numbers of pathogenic microorganisms. Furthermore, these organisms are not part of normal soil microflora. Notable exceptions are the sporeforming organisms such as those causing tetanus, gas gangrene, and anthrax. Despite their incompatability, an investigation of 228 reported outbreaks of waterborne disease (25,984 cases) between 1946 and 1960 (Weibel, et al., 1964) indicated that, of the outbreaks attributable to the quality of raw surface or ground water, many were potentially attributable to passage of pathogens over or through soils. However, many years of experience with agricultural utilization of sewage sludge and effluent throughout the world have demonstrated that contamination can be prevented through proper waste management procedures.

Survival in Soil

Data on survival of human pathogens and indicator organisms in soil are abundant. However, they are highly contradictory. Probably this diversity of findings is due to the variety of conditions in the soil environment which influence longevity of pathogens in soils and to the variety of experimental conditions under which observations have been made. In a review paper on the subject, Rudolfs, et al. (1950), noted that the viability of Salmonella typhosa, the typhoid bacillus, has been reported to vary from less than 24 hours in peat to more than 2 years in freezing moist soils, but was normally less than 100 days. On the basis of their examination of available data, they reached the general conclusions that longevity of S. typhosa in soils was favored by low temperatures, high moisture concentrations and high moisture holding capacities. They also reported that sunlight and low pH values were unfavorable to S. typhosa in soil, and the presence of sewage or feces in soil either reduced or had no effect on the survival of typhoid bacteria.

The fate of coliform organisms in soils is of interest because of their widespread use as indicators of the possible presence of pathogens. Rudolfs, et al. (1950), reviewed available information on survival of Escherichia coli in natural soils and concluded that they survive for longer periods than S. typhosa. Young and Greenfield (1923), for example, found that E. Coli survived for 5 years in a soil sample. Rudolfs, et al., suggested that "to await the death of all or most E. coli before considering a soil no longer capable of carrying disease organisms may prove to be too extreme a test." Geldreich, et al. (1962), however, considered fecal coliforms and total coliforms, to a lesser extent, to be indicative of probable human or animal contamination of soils. A study of 251 unpolluted soil samples indicated an absence of appreciable numbers of fecal coliforms.

Van Donsel, et al. (1967), found that fecal coliforms survived at the surface of soils longer than fecal streptococcus during the summer, but that fecal streptococcus survived much longer than fecal coliforms during the spring and winter. The time required for 90 percent reduction of fecal coliforms in soils ranged from 3.3 days in the summer to 13.4 days in the fall. The 90 percent reduction times for fecal streptococcus ranged from 2.7 days in the summer to 20.1 days in the winter.

Van Donsel, et al. (1967), noted that aftergrowth of coliforms, fecal coliforms, and fecal streptococci can occur in soils following their contamination. In such cases, the soils could yield concentrations of indicator organisms which are out of proportion to their actual sanitary history. It was concluded that the repeated aftergrowth of total coliforms at the surface of soils following rains in warm weather rendered that broad group of indicator organisms unsuitable as indicators of soil contamination.

In summary it may be said that while soil is not a favored environment for pathogenic microorganisms, survival for appreciable periods is possible. Common indicator organisms may at times give an overly severe appraisal of the bacteriological quality of soils.

Movement With Surface Water

In addition to longevity of organisms in soil, their movement through and over the soil is of interest. Reports on the bacteriological quality of storm water runoff suggests that movement of organisms from soils to surface waters is of concern (U.S. Public Health Service Report, 1964), but little quantitative information is available. Geldreich (1966) noted that storm drainage was characterized by the predominance of organisms indicative of warmblooded animal pollution rather than by indicators of human pollution. Van Donsel, et al. (1967), who studied the densities of strains of fecal coliforms and fecal streptococcus in runoff from plots innoculated with the organisms found that the tracer organisms were most readily isolated in the spring during periods of prolonged heavy rainfall. Once the soil dried following prolonged rains, subsequent showers did not result in the organisms being present in the runoff water.

In investigating the effect of agricultural drainage on water quality, Henderson (1962) found that the coliform density of the Potomac River at Great Falls increased from an average of 225 to 35,000 per 100 ml following rainfall. BOD measurements on wet-weather flushing flow conditions on animal drainage sheds showed that the oxygen demand on streams during rainfalls exceeded the population equivalent of the animals, and thus reflected flushing of stored wastes into the stream. Hence, proceedures for land application of wastes must consider the possibility of surface water contamination.

Movement With Ground Water

Because of the large number of pit privies, septic tank percolation fields, and other opportunities for contamination of ground waters with pathogenic organisms, there has long been interest in the movement of bacteria through the soil. Many reports have given observed conditions at various locations. Unfortunately, it has not lead to a body of fundamental knowledge on which the movement of bacteria downward by percolation in unsaturated soil or laterally in aquifers can be readily predicted.

McGauhey and Krone (1967) summarized a dozen reported observations of travel of bacteria in the soil. Travel distances ranging from 10 to 800 feet were reported. McGauhey and Krone concluded that coliforms and other bacteria move but a few feet with percolating water in unsaturated flow and a few hundred feet in ground water in saturated systems. On the basis of another survey, Mallmann and Mack (1961) made the following conclusions regarding bacterial travel from recharge wells receiving diluted sewage effluent: bacterial pollution from recharge wells will travel only in the direction of ground water flow, and then seldom over 100 ft. immediately after start of recharge operations. As recharge continues, particulate matter in the recharge water deposits near the source of recharge to form a filtering media which is effective in removing bacteria. As the filtering efficiency is improved, the extent of bacterial travel retreats to within a few feet of the well. Mallmann and Mack indicated that absorption and filtration in the aquifer accounted for removal of bacteria.

Studies of land disposal of waste treatment plant effluents at Whittier and Azusa, California (Stone, 1953) demonstrated that "a well-oxidized effluent meeting bacteriological drinking-water standards can be produced after percolating sewage effluent through 3 to 7 ft. of surface soil". McGauhey and Krone (1967) cited Rumanian experience which indicated 92 to 97 percent removal of coliforms in the top 1 cm of soil with an additional 3 to 5 percent being retained in the next 4 cm. Baars (1961) noted experience in the Netherlands which indicated complete removal of *E. coli* by filtration through 5 meters of dune sand. He concluded that failure of soil systems is normally noted first by the presence of chemical pollutants and that bacterial pollution of ground waters represents a later phase of pollution which is generally accompanied by anaerobic conditions in the soil.

Extensive studies on the fundamental mechanisms involved in movement of coliforms in porous media were summarized by Krone, *et al.* (1958). Two phases of bacterial removal were noted---rapid removal occurred at the surface and as the flow progressed into the soil, continued removal occurred in proportion to the number of remaining bacteria. Removal was thought to occur by two mechanisms. One was mechanical straining and the other was sedimentation on soil surfaces within the pores of the media. Krone, *et al.*, showed that as water bearing *E. coli* was applied to columns of soil, an abrupt breakthough of bacteria, corresponding to the arrival of chemical contaminants occurred. Following the initial breakthrough, accumulation of material at the upper surface of the soil improved straining efficiency and resulted in reduction of the number of bacteria in the column effluent. As straining sites on the surface became clogged, clusters of organisms were sheared from the upper layer of soil and carried deeper into the column. These clusters which are sloughed off are then removed deeper in the soil by straining and sedimentation. Ultimately, virtually all of the bacteria applied to soil are removed by straining or sedimentation, or their movement is sufficiently retarded so that die-off occurs.

These general concepts applied to both saturated and unsaturated columns, but bacterial removal occurred at slower rates with the latter.

The findings of Krone, et al., are compatible with the studies previously described and those of Stiles, et al. (1927), and Caldwell (1938) regarding the retreat of the front of bacterial penetration after a period of time. However, McGauhey and Krone (1967) have shown that the clogging of straining sites which occurs at the surface is repaired by a period of resting to allow oxidation under aerobic conditions. Hence, part of the bacterial removal efficiency of soil systems might be temporarily reduced between subsequent sludge applications.

In summary of the problem of travel of pathogens through soil, it is concluded that all bacteria will be removed ultimately by the media. While data on travel of bacteria in soil as a result of sludge application are lacking, it would seem that it is the maximum travel distance following the initial period of application which should be of concern. Then, due to clogging of soil voids, the front will regress to an "equilibrium" position where the rate of addition by sludge application and any replication in the soil is equal to the bacterial removal by death. Hence, the problem of bacterial contamination of ground water is minimal so long as ground water is withdrawn at a sufficient distance from the application area.

5.1.2 Viruses

The fate of viruses when applied to soil has been investigated by several workers in laboratory systems. Bacteriophage T_2 and poliovirus type 1 (Sabin) were found to adsorb reversibly (> 90%) onto clays like Illite, Kaolinite, Montmorillonite and naturally occurring Missouri River clay in the presence of cations like sodium, potassium, and aluminum (Carlson, et al., 1968). However, the efficiency was reduced considerably when albumin or sewage was present in the system.

The fate of type 1 poliovirus in water moving slowly through sand was studied in the laboratory by Robeck (1966). With feed containing 10,000 - 30,000 viruses/ml, and a 2 ft. bed of sand, with a flow rate of 3 - 4 ft/day, the first appearance of 1 - 12 virus units in the effluent did not occur until 6 weeks in

Chillicothe sand 15 weeks in Newton sand.

The conclusions drawn from special percolation tests at the Santee Recreation Project (1967) indicated that attenuated polio type 3 virus was effectively removed from effluents at distances of 200 to 1500 ft. of subsurface percolation.

On the basis of the results obtained in their column studies, Drewry and Eliasen (1967) observed:

> "Virus movement through soils under saturated conditions should present no great hazard with respect to underground water supplies. This would, of course, be limited to continuous strata of common soils. . . . "

Cases involving passage of infectious hepatitis and other viruses through soils have been reported in the literature (Hallgren, 1942; Paczenik, et al., 1956; Weber, 1959; and O'Connor and Morris, 1955). However, in most of the cases where virus appeared to have travelled in the water through the soil, evidence indicates that channeling through fissured or fractured substrate had occurred rather than seepage through the soil itself (Drewry and Eliasen, 1967). There is one published report of the failure of soil columns (garden soil) in the laboratory in removing viruses (Gilcreas and Kelly, 1955).

It may be concluded, on the basis of the information available in the literature concerning the persistence of viruses in anaerobically digested sludge and the fate and movement of viruses through sand and soil, that land application of digested sludge is unlikely to create a serious virus hazard. Although it is possible that anaerobically digested sludges may contain very low concentrations of viruses which cannot be detected by the presently available methods, percolation through ground formations of continuous strata would be expected to effectively remove those viruses.

5.1.3 Chemical Additions

The influence of waste substances on soil and crops is considered in Section 5.2. Here, those chemical substances which may produce a notable effect on the quality of ground or surface waters are discussed.

The fate of chemicals added to soils depends on their nature and the characteristics of the soil. Some chemicals may pass freely, others may be tightly held, and some may undergo alteration or degradation in the soil. In cases where substances are removed in passage through soil, the capacity of the soil system for continued removal with sustained application is of concern. In cases where the chemical nature of waste constituents is altered by passage through soil, the fate of the altered form in the soil system is of concern.

The nature of many biological transformations in soil depends upon whether aerobic or anaerobic conditions prevail. Carbon, nitrogen and sulfur compounds are principal examples of substances whose form is highly dependent on the oxidation level in the soil. The type of soil, moisture content, and amount of biological demand for oxygen are significant factors influencing aeration of soils.

Results of studies of waste water application to soil reviewed by McGauhey and Krone (1967) illustrate the types of interaction of soil systems with constituents of percolating water. Under aerobic conditions, chloride, nitrate, sulfate, bicarbonate, calcium, magnesium, sodium, and potassium ions move freely through the soil. Biological oxidation in the upper levels of soil results in reducing biodegradable organics and increasing the concentration of sulfate; bicarbonate, and nitrate ions. Phosphorus and ammonia disappear in the upper layers of the soil. In contrast, where anaerobic conditions prevail, nitrates and sulfates disappear in the upper layers of soil as a result of biological reduction.

A review of the significance of pollution by chemicals in waste has been presented by Hanks (1967). Of the many possible pollutants to the water environment as a result of waste application to soil, nitrogen, phosphorus, and heavy metals appear to be of major concern and are given special consideration here.

Significance of Nitrogen

As indicated in Appendix A, the total nitrogen content of a digested sludge slurry of 3.0 to 3.5% concentration is about 1500 mg/l. Neglecting losses, irrigation with one inch of this sludge would add a total of about 340/lb acre of nitrogen to the land. About 150 lb. of this nitrogen would be in solution (principally as ammonium nitrogen), and the remaining 190 lb. would be contained in the solid phase. The total exceeds the normal annual rate of nitrogen addition in commercial fertilizers to heavily fertilized soil (Boone, 1969). Thus the fate of nitrogen following sludge application to soil is of concern.

As described in sections which follow, nitrogen in wastes may be biologically converted to nitrates which travel freely through soil. High nitrate concentrations in water may cause methemoglobinemia, an infant disease characterized by a reduction in the oxygen carrying power of the blood. Literature relating to incidences of methemoglobinemia due to nitrates in water has been reviewed by Walton (1951). The U. S. Public Health Service Drinking Water Standards (1962) recommend that nitrates be limited to 45 mg/l as NO_3 (10 mg/l as N) in drinking water.

High nitrate concentrations may also be of concern in waters used for stock and wildlife watering. Burden (1961) has reported illness and death of farm animals as a result of consumption of water with high nitrate concentrations.

Nitrates may be found in ground waters in high concentration due to leaching through soils receiving nitrogen bearing wastes or chemical fertilizers. They may occur in surface waters because of surface runoff, tile drainage, or base flow originating from the same soil. It is likely, however, that some of the nitrates in surface waters would be converted to organic nitrogen through biological synthesis. Just as nitrogen serves as nutrient for plant life in soil, so it can fertilize plant life (notably algae) in surface waters. However, under nitrogen limiting conditions, algal synthesis can still occur because of the ability of some blue green algae to fix nitrogen from the atmosphere (Sawyer, 1952).

Transformation and Fate of Nitrogen in Soil

McCarty (1967) has indicated that the average nitrogen content of soils in the United States to plow depth (6 in) is 3000 lb/acre. Most of this is in the form of stable organic or complexed compounds.

Nitrogen, as it is added to soil in waste products, would normally exist as organic nitrogen or ammonium ions. In the soil this nitrogen can undergo many transformations as a result of biological activity. Organic compounds containing nitrogen are enzymatically hydrolyzed to amino acids which are microbiologically deaminated to produce ammonia (which at normal pH values will exist as the ammonium ion). This overall biological process of converting organic nitrogen to inorganic nitrogen is called mineralization. The rate of conversion may not be much greater than that occuring with soil organic matter. Evidence of the slow degradation of sludge organic matter was presented by Mann and Barnes (1957). They found that eighty percent of the organic matter added over a period of eight years was retained after the ninth year.

Ammonia in wastes applied to soil or produced by mineralization of organic nitrogen in the wastes undergoes further biological transformation. Under aerobic conditions, ammonia is biologically oxidized to nitrate in a process called nitrification. Nitrification results from oxidation of ammonia to nitrite by the microorganism Nitrosomonas followed by oxidation of nitrite to nitrate by another bacteria, Nitrobacter. The overall rate of mineralization and nitrification of nitrogen in digested sludge solids has been reported by Anderson (1956). He found that only 20 percent of the nitrogen was converted to the nitrate form within 4 months. However, this rate is highly dependent on factors such as pH, temperature and moisture.

Other microbiological transformations can occur under anaerobic conditions, such as may exist in flooded soils and/or in soils with an extremely high oxygen demand as a result of biological decomposition. To a lesser extent, anaerobic conditions may exist in microenvironments in aerated soils. Under these reducing conditions, nitrates serve as electron acceptors in oxidation of organic material and are, in the process, reduced to ammonia, free nitrogen, or nitrous oxide.

The reverse of mineralization may also occur in soil as a result of synthesis by plants or by microorganisms growing in the soil. This process is called immobilization. Most plants require aerobic conditions, but immobilization by bacteria can occur under anaerobic conditions.

Recently, other, nonbiological, mechanisms for nitrogen loss from soil have been investigated. Allison (1966) has summarized that "these other mechanisms involve the reaction of nitrous acid or nitrates with certain soil constituents, but there is no certainty as to exact reactions that take place. Microorganisms are not involved except that they produce the reacting substances". Allison concluded that these mechanisms may produce large nitrogen losses where heavy application of reduced nitrogen is involved. Hence, they may be of importance in sludge application to soil.

All of these nitrogen transformations in soil are important because they determine the fate of nitrogen applied to soil. Nitrogen synthesized into plant protein can be harvested as a useful product. Nitrogen converted to free nitrogen or nitrous oxide is lost to the atmosphere. Additional nitrogen will be lost through volatilization of ammonia. The movement of nitrogen from soil into the water environment depends on the form in which it exists.

Nitrate is the mobile form of nitrogen in soils. It is highly soluble, is not adsorbed on soil and hence moves with soil water. Ammonia nitrogen is readily adsorbed on soils, and does not move with the soil water. The readiness by which adsorbed ammonium can be replaced by other percolating cations can become reduced by the phenomenon of ammonium fixation (Grim, 1953). The mobility of organic forms of nitrogen in soil is limited by insolubility and/or adsorption.

The validity of these generalizations regarding the mobility of nitrogen forms in soil are confirmed by studies summarized by Bailey (1968). In general, over 99 percent of the nitrogen reported to be found in leachate has been in the nitrate form. Recent findings of Preul and Schroepfer (1968) on the movement of nitrogen from wastes in soil also confirm that adsorption of ammonia controls movement and that nitrogen in the percolating water is in the nitrate form.

In summary, nitrogen in wastes applied to soil will be in the ammonia form or will be biologically converted to the ammonia form. Leaching of nitrogen in this form is not expected. However, when ammonia nitrogen is converted to nitrate nitrogen as a result of biological oxidation, this form will readily pass through soil with the percolating water. Some nitrogen will be immobilized in the form of microbial cells in the soil, but nitrogen will be recycled over a period of time. Nitrogen used by plants will be converted to organic nitrogen and can be removed by harvesting. Some nitrogen will be lost to the atmosphere by volatilization of ammonia and by loss of nitrogen gas or nitrous oxide produced by biological denitrification or other mechanisms.

Nitrogen Balance

Ideally, the amount of nitrogen applied to soil in the form of wastes would equal that lost from soil in the form of plant protein plus the gaseous losses. In this section nitrogen sources and losses are considered in order to evaluate the possible effect of waste application on the water environment. It must be emphasized that insufficient data is presently available for making a nitrogen balance under heavy sludge loadings. Therefore, the discussion serves only as an indication of the magnitude of the problem.

As given previously, an inch application of digested sludge would add 340 pounds of additional nitrogen to the soil. Precipitation may add another 5 lb/acre/yr (Miller, 1955), but this will be ignored. Likewise, nonsymbiotic nitrogen fixation will be ignored (Allison, 1966). Therefore, the total annual input will be taken as 150 lbs. of ammonium nitrogen and 190 lbs. of organic nitrogen. If one assumes a mineralization of five percent of the organic nitrogen during the growing season, then 160 lbs. of available nitrogen will be added to the soil.

It is difficult to quantify the losses of nitrogen from soil. The amount removed by harvesting is dependent upon agricultural practices. A survey of recent literature would suggest that 150 lb/acre/yr as a reasonable approximation for the amount of nitrogen which can be removed as plant material under modern agricultural practices.

Information on nitrogen loss through evolution of nitrogen gas and nitrous oxide and volatilization of ammonia has been reviewed by Allison (1966). He summarized that in tests where no leaching occurred, "only rarely have carefully conducted vegetative tests shown nitrogen recoveries in the crop plus soil greater than about 95 percent of the applied nitrogen; values of only 70 to 90 percent are fairly common, and a few are as low as 60 percent."

Experiences with application of animal manures indicate that significant losses of nitrogen through volatilization of ammonia can occur. In one series of tests, 42 percent of the applied nitrogen was lost in four days, except where the animal manure was plowed under (Iverson, 1932). Ammonia losses of 30 to 40% with the application of urea have also been reported (Volk, 1961).

Losses of nitrogen under heavy sludge loadings may be much greater than indicated by prior studies. The additional water,organic matter and heavy metals may all increase chemical nitrogen losses. For purposes of illustration, losses to the atmosphere will be taken as 50 percent of the amount of available nitrogen applied. An allowance for nitrogen loss due to immobilization and accumulation of "nonbiodegradable" organic nitrogen compounds will not be made in this approximate nitrogen balance. The role which these mechanisms would play in soils receiving heavy waste loadings is largely unknown. In ignoring the accumulation of organic nitrogen, it is tacitly assumed that an equilibrium condition will be reached where the rate of mineralization equals the rate of accumulation of organic nitrogen.

Likewise, for purposes of considering the effect of waste application on ground water, loss of nitrogen through surface runoff will be ignored. It should be noted, however that a nitrogen balance for harvested crop area in the United States in 1930 tabulated by Allison (1955) indicated surface losses as approximately equaling losses through crops and losses by leaching. Also, Harmeson and Larson (1969) found that high nitrate concentrations in rivers draining agricultural areas corresponded to periods of high discharge. Mackenthum, et al. (1964), found nitrogen loss by overland drainage in the Madison, Wisconsin area to be in the range from 4.4 to 6.4 lb/acre/yr. Mackenthum, et al. (1964), also reported results of Eck and his co-workers who found that nitrogen loss depended on slope of the soil. Loss from land on 20 percent slopes was 38 lb N/acre/yr and this was reduced to 18 lb N/acre/yr on an 8 percent slope. Clearly, the significance of nitrogen loss by surface runoff cannot be ignored in management of waste disposal system. It is merely ignored here to facilitate estimation of potential effects on ground waters.

The total available nitrogen loss is therefore assumed to be 50 percent of the ammonium nitrogen added plus 150 lb/acre/yr loss in form of plant protein. Considering the annual available nitrogen addition (in lb/acre) to be 160 X where X is the digested sludge slurry application rate in inches/yr., the net excess available nitrogen will be approximately (80 X - 150) lb/acre/yr. Hence with sludge application rates equal or greater than two inches/yr., more available nitrogen will be added to the soil than will be lost as crops or gas. In the absence of measures to prevent excessive nitrogen loss, the excess nitrogen will be oxidized to the nitrate form and carried through the soil with percolating water and the concentration of nitrates in leachate can be approximated. An average concentration is computed, although it is recognized that seasonal variations will occur. To allow estimation of nitrogen concentration, assume that 25 percent of rainfall on cropped land percolates through the ground (Collison and Mensching, 1930). A 40 in/yr rainfall will be considered. For convenience, the same fraction of liquid from sludge will be considered to percolate downward. Hence, the amount of leachate will be 0.25 (40 + X) in/yr. With appropriate dimensional conversions, the concentration of nitrogen in the percolate will be (80 X - 150) / (2.27 + 0.057 X) in mg/l. The table which follows shows leachate nitrogen concentrations for various sludge application rates calculated in the manner described above.

Projected Nitrogen Concentrations in Leachate

Sludge Application Rate (in/yr)	Total Available Nitrogen Application (lbs/acre/yr)	Total Excess Available Nitrogen (lbs/acre/yr)	Projected Nitrogen Conc. in Leachate (mg/l as N)
2	320	10	5
3	480	90	40
4	640	170	70
6	960	330	130
8	1280	490	180
10	1600	650	230
12	1920	810	280

Few data are available to substantiate these projections because nitrogen application rates of the magnitude shown in the table are uncommon. Stauffer (1942) reported on lysimeter studies in which nitrogen concentrations in leachate were as high as 71 mg/l as N. Welch and Bomke (1969) reported nitrogen concentrations in central Illinois field tile discharges in the order of 15 mg/l at N. Harmeson and Larson (1969) report that the nitrate content of the entire Kaskaskia River in central Illinois averaged over 6 mg/l as N during the summer months and was sometimes as high as 13 mg/l.

It must be reiterated that these calculations are based on several **arbitrary** assumptions. Nitrogen losses could conceivably be much greater than allowed for in this example. Furthermore, as discussed in Section 7.3, several nitrogen removal techniques are possible before application to the land. Also the nitrogen content of most liquid sludges may be considerably lower than the 1500 mg/l assumed. Few thorough analyses have been made on the total sludge and the value taken here is based mainly on Chicago data. However, recalling that the U.S. Public Health Service Drinking Water Standards suggest a limit of 10 mg/l on nitrogen, the preceeding table indicates that leaching of nitrogen from soil to water is a major consideration in management of waste disposal systems.

Phosphorus

The fate of phosphorus in wastes applied to land is of concern because it serves as a nutrient for algae growths if it is permitted to reach the water environment. Although many nutrient materials must be present in sufficient quantity to stimulate growth of algae, phosphorus is the nutrient most frequently considered to be limiting in the normal aquatic environment (Sawyer, 1952 and Ohle, 1953). In his work on lakes at Madison, Wisconsin, Sawyer (1947) indicated that phosphorus concentrations in excess of 0.01 mg/l could be expected to produce nuisance conditions due to algae. Concern over the influence of phosphorus in industrial and domestic wastes on eutrophication has lead to recent requirements, notably in the Great Lakes area, for phosphorus removal from conventional waste treatment plant effluents.

Phosphorus behaves quite differently from nitrogen in soils. It is readily fixed in soils and does not migrate with soil water. The mechanisms of phosphorus fixation and the factors influencing the phosphorus fixing ability of soils have been reviewed by Bailey (1968). This inability of phosphorus to travel through soils has been confirmed by several studies of waste application on land. For example, in the Penn State studies of waste water renovation, 99 percent removal of phosphorus from percolating water was observed (Parizel, et al., 1967).

While massive ground water contamination with phosphorus is not of immediate concern, the capacity of soil for fixing phosphorus and rate of accumulation of phosphorus through waste addition is of interest. Considering liquid digested sludge to have 800 mg/l of phosphorus, application of one inch of sludge will bring 180 lb/acre of phosphorus to the soil. Considering the phosphorus content of plant matter to be 10 percent of its nitrogen content, about 15 lb/acre/yr of phosphorus will be harvested annually. Hence, the annual phosphorus accumulation in lb P/acre will be 180X - 15 where X is the total annual sludge application rate in inches.

The ability of soil to fix this annual accumulation of phosphorus depends on the type of soil, particle size, pH, oxidation-reduction potential, temperature and organic content. These factors have been reviewed by Bailey (1968). Several studies reported in the literature serve to give some idea of the capacity of soils to fix phosphorus. Morgan and Jacobson (1942) applied a total 3005 lb/acre of phosphorus over an 11-year period. The average amount of phosphorus which leached beyond the 18 inch depth was 0.1 lb/yr. Bailey (1968) tabulated data from Toth and Bear which indicated the "phosphorus absorbing power" of three New Jersey soils to be from 14 to 125 tons of 20 percent superphosphate per acre. Bailey (1968) indicated that this capacity for fixing phosphorus is largely in the upper horizons of the soil. The average phosphorus content of the upper 6 2/3 inches of various Iowa soils was listed by Millar (1955). Sandy loams contained 864 lb/acre; loams, 1205 lb/acre; silt loams, 1288 lb/acre and silty clay loams, 3089 lb/acre.

While large quantities of phosphorus are not expected to reach ground waters as long as the phosphorus fixing capacity of the soil is not exceeded, it may reach surface waters as a result of storm runoff. Because of the reaction of phosphorus with soil, the amount of phosphorus carried away in the soluble form by surface runoff is limited. McCarty, et al. (1967), indicated that the concentration of soluble phosphorus in water in equilibrium with soil would be about 0.25 mg/l. However, additional phosphorus may be carried by runoff in the precipitated form or adsorbed to soil particles. Weibel, et al. (1966) found an average of 0.6 mg/l of phosphorus in drainage from a winter wheat field. Sylvester (1961) found the total phosphorus content of surface drainage waters to be 0.25 mg/l of which 0.09 mg/l was suspended.

Although these concentrations of phosphorus in surface drainage are significant with regard to the quality of receiving waters, loss of phosphorus by runoff has little effect on the phosphorus balance of the soil. Engelbrecht and Morgan (1961) who studied phosphorus in runoff water in the Kaskaskia

River basin in Illinois, reported that while individual samples showed rates of phosphorus loss of up to 15 lb/acre/yr, the mean of all samples was 0.35 lb/acre/yr. This result was in close agreement with Sawyer's (1947) findings which showed that agricultural land drainage near Madison, Wisconsin contained 0.4 lb P/acre/yr.

In summary, phosphorus is tightly held by soil, and movement through the soil into ground water as a result of waste application to soil is not expected to occur in high concentrations. However, the phosphorus fixing capabilities of soil are not unlimited, and saturation and ultimate breakthrough of phosphorus is possible from land disposal sites operated over an extremely long period of time. Loss of phosphorus in surface drainage in quantities sufficient to impair receiving water quality is possible, and waste application and management practices must take account of this eventuality.

Heavy Metals

The U. S. Public Health Service Drinking Water Standards (1962) contain mandatory limits on cadmium (0.01 mg/l), hexavalent chromium (0.05 mg/l), lead (0.05 mg/l), and silver (0.05 mg/l). The restriction on silver is principally to prevent its indiscriminant use as a disinfectant. Cadmium, and lead are cumulative poisons, that is, when injested in low concentrations, they accumulate in the body and may manifest ill effects only after a long period of consumption. Other heavy metals - copper and zinc - are included in the non-mandatory section of the Drinking Water Standards, but not principally for health reasons. The suggested limit on copper is 1 mg/l because of its effect on taste, and the suggested limit on zinc is 5 mg/l for the same reason. Heavy metals in wastes applied to soil are also of concern because of the possibility that they may be toxic to plants. The significance of heavy metals in that regard is considered in Section 5.2.3.

While few data are available on the potential significance of heavy metals in sludges applied to land on water quality, leaching of heavy metals is not expected to constitute a serious hazard. Jenkins and Cooper (1964) were able to leach considerable amounts of heavy metals from dried digested sludge by percolating water through it. However, when the sludge was mixed with soil in proportions to be expected if the sludge was used agriculturally, only traces of the metals could be leached by percolating water.

Possible mechanisms which control the migration of heavy metals in soils recently have been reviewed by Jenne (1968). He proposed that the principal control of the concentrations of heavy metals in soils and waters is their adsorption on hydrous oxides of manganese and iron which are "nearly ubiquitous" coatings in clays, soils and sediments. Of the factors reviewed by Jenne, influencing occlusion of heavy metals in this fashion, pH was indicated as being one of the most significant. At low pH values heavy metal adsorption is less efficient because of interference with the hydrous oxides of manganese and iron.

In summary, the health significance of heavy metals demands that their fate in the environment following application of wastes to land be considered. Results of field experiments with various soil types, heavy metal concentrations, pH conditions, oxidation reduction potentials, and relating and complexing conditions are not available to permit assessment of the fate of heavy metals in the soil. However, available information suggests that they will be fixed in the soil to the extent they will not adversely influence water quality. As with phosphorus, the capacity of the soil for continued removal of heavy metals over an extremely long period of application of high concentrations needs to be considered.

5.2 EFFECT ON SOILS AND CROPS

Sewage and sewage sludges have been widely used in agriculture throughout the world for many years. In Europe, the Union of South Africa and the United Kingdom, intensive applications have been developed. It was estimated in 1954 that 800,000 tons of dry sludge solids were produced in England and Wales. Of these, 350,000 tons (43.75%) were applied to land in various forms as manure (Stone, 1962).

The extent to which the disposal of liquid sewage sludge on land is practiced at the present time in the United States is not known. Scott (1962) and Dean (1968) document the growing adoption of the practice, but many more treatment plants than they mention are known to be using land disposal of liquid sludge. Most of the treatment plants are small. The systems for disposal of liquid sludges on land have generally been simple, with little thought given to obtaining quantitative measurements.

5.2.1 Chemical Additions

Normal plant growth and development requires sufficient nutrients in proper balance. Of the sixteen chemical elements that are known to be essential for green plants, the major or macronutrients are nitrogen, phosphorous and potassium. These are the nutrients taken in from the soil in the greatest amount. Secondary nutrients are calcium, magnesium and sulfur. Micronutrients include iron, manganese, copper, zinc, boron, molybdenum and chlorine. Typical element concentrations for digested sludge are shown in Table 5.2.1.

Macronutrients

The nitrogen, phosphorus, and potassium fertilizing capacity of sludges have been the subject of a number of reports. Much of this work has been done with dried sludges which lost some of their ammonium nitrogen during the drying process. If sludges are to be used as a source of plant nutrients, it is advantageous to apply digested sludge as the original 3 to 5 percent suspension. If the ammonium nitrogen is lost by drying, digested sludge is a poor nitrogen fertilizer because mineralization of organic nitrogen

TABLE 5.2.1

TYPICAL ELEMENT CONCENTRATIONS AND PROPERTIES FOR DIGESTED SLUDGE(1)

Element	mg./l.	lbs/ac. -in.	Element	mg./l.	lbs/ac. -in.
Total Nitrogen	1500	340	Zinc	110	25
NH_3 Nitrogen	660	150	Boron	4	1
Phosphorus	800	180	Molybdenum	0.09	0.02
Potassium	150	34	Chlorine	290	65
Calcium	1210	275	Lead	70	16
Magnesium	175	40	Chromium	50	13
Sulfur	90	20	Nickel	4	1
Iron	1270	290	Cadmium	3	0.7
Manganese	40	9			
Copper	30	7			

pH	7.45
Conductivity (μmhos/cam)	4,800
Specific gravity	1,007
Maximum particle size (mesh)	10.0
Total solids (lbs/ac. -in)	6,760
Mineral solids (lbs/ac. -in)	3,610
Organic solids (lbs/ac. -in)	3,150
Hexane soluble (lbs/ac. -in)	675
5 Day BOD(2) (mg/l)	3,000

(1) Unpublished date, Hinesly & others.

(2) Scott (1962).

is very slow.

Studies conducted with digested sludge slurries have generally confirmed the expectation of adequate amounts of available nitrogen and phosphorus. In fact, the nitrogen content of the fresh dewatered digested sludge used by Lunt (1959) prompted him to suggest that rates of sludge application be governed by the amount of nitrogen they supply rather than by the amount of organic matter applied. He found heavy applications provided too much nitrogen for a balance of plant nutrients. In comparison, digested sludge has a rather low content of potassium. Because of its high solubility, most of the potassium is lost with the treatment plant effluent.

Secondary and Micronutrients

The *calcium* and *magnesium* supplied in a one-inch application of digested sludge are about what is needed to replace the annual loss due to leaching and crop removal. The *sulfur* supplied by digested sludge is more than adequate. Sulfur is seldom deficient in soils but if it were, digested sludge would supply it in about the proper ratio with the available nitrogen for most cultivated plants.

From Table 5.2.1, it may be seen that many of the essential minor elements are present in digested sludge in seemingly minute amounts. But, because many of these can accumulate in soils, what appears to be harmless amounts may increase to injurious levels. The margin between deficient and toxic levels of essential minor elements is small.

Iron content of normal soils ranges from 0.5 to 5 percent. Iron is not generally considered as a soil contaminate. Toxic concentrations of iron have not been reported except at low pH and high moisture levels in soils. Deficiencies of iron, especially in the presence of excessive manganese, are more often reported for normal soils than are toxic levels.

Manganese is normally found in soils at levels of 0.01 to 0.5 percent. Manganese contamination of soils are reported to have occurred from mine seepage and from applications of impure phosphate fertilizers, but the major concern of most investigators has been manganese deficiencies rather than toxicity. Manganese is likely to be distributed throughout the soil profile. If soils should be contaminated with manganese, they can be readily reclaimed by liming to a neutral pH.

Copper concentrations range from 2 to 50 PPM in normal soils. Copper appears to be rendered immobile in soils by some mechanism of adsorption by silicate minerals and complexing and chelation by organic matter. There is a very strong tendency for copper to accumulate in the soil surface. Shallow rooted crops would show copper toxicity symptoms first, but with continued applications of copper, toxic levels might eventually be reached for all crops. Liming and the addition of zinc and manganese have sometimes counteracted the injury caused by copper contamination of soils.

Zinc addition limits are difficult to establish because like copper it is bound strongly by many soils. Generally, zinc deficiencies have been of more concern than have excess concentrations. In fact, concern has been expressed with regard to the insufficient supply of zinc in diets that depend mainly on plant rather than animal protein. Zinc concentrations in soils vary from 10 to 250 PPM. Toxicity symptoms due to zinc have been alleviated by liming acid soils, adding iron chelates or, in some cases, nutrient solutions containing copper salts.

Boron concentrations in soil range generally from 20 to 200 PPM on a dry weight basis. The difference between a deficient and toxic concentration in soils is very small. Even though large amounts of boron may be discharged with detergents to the waste treatment plant, very small concentrations of boron are contained in digested sludge. Evidently most of the boron is lost in the effluent.

Molybdenum requirements of most plants are very small. In cultural solutions as little as 1 part of molybdate in a billion parts of solution has been found to be adequate for most plants. On the other hand, no injury is exhibited by plants having accumulated molybdenum in tissues in excess of 100 PPM on a dry weight basis. Yet, the data indicates that amounts in excess of 20 PPM in plant tissue are nearly always associated with disturbed copper metabolism in ruminant animals. Unlike the other essential minor elements, molybdenum availability and uptake is increased by liming an acid soil. Digested sludge would supply only about 0.02 lbs/acre of molybdenum per inch application but this is probably adequate for plants on non-acid soils. Soils normally contain from 0.2 to 5 PPM of molybdenum.

Chlorine is an essential nutrient for only some plants. The quantity required is so low that a deficiency of the element never develops in nature. Most soils contain about 100 PPM of chlorine, but chlorine is very mobile in soils and its concentration can change rapidly with rainfall. In general, chlorine is not toxic to plants as some plants materials contain high amounts of chlorine.

Non-essential Elements

Lead is a cumulative poison. Therefore continual consumption of small quantities, that by themselves cause no harm, may eventually reach toxic levels. Lead concentrations in soils frequently range from 2 to 200 PPM. The rate of deposition on soils and the uptake of lead by plants growing thereon has been conclusively shown to be a function of the area's proximity to highways and urban areas (Lagerwerff, 1967). Plants may take up to 10 PPM of lead on a dry weight basis before showing toxic symptoms while growing in cultural solutions. It is not uncommon for plants growing on highly contaminated soils to accumulate lead to the extent of 40 to 50 PPM of their dry weight.

Chromium is taken up by plants in small concentrations. Although it is extremely important from the standpoint of animal diets, there is no evidence that chromium is of use to plants. Even though soils normally contain only 1 PPM of chromium, it is not very toxic to plants when found in greater concentrations, except at a low pH. Chromium in the form of chromate is particularly toxic. but this form would be stable only under alkaline, oxidizing situations. Digested sludge application on land could significantly increase the level of total chromium in soils and perhaps, with repeated applications, in time increase the available or soluble chromium to a toxic level.

Nickel is found in rather small amounts in digested sludge. However, it may accumulate in soil surfaces and be suspect as a soil contaminate with continued applications. The nickel content of most soils ranges from 2 to 40 PPM. It is not considered to be essential for plant or animal nutrition, but it is absorbed by plants normally in concentrations of 0.1 to 5 PPM of dry tissue. At concentrations exceeding 50 PPM in plant tissue, toxic symptoms are exhibited and plant growth is decreased. Application of iron may be used to counteract nickel toxicity, but liming is more practical. Nickel is not likely to be a problem except in acid soils.

Cadmium concentrations in soil normally range from 0.01 to 7 PPM. This element can be absorbed by a large number of plant species in concentrations that may exceed 3 PPM before severe plant depression occurs. Cadmium is of concern because it will accumulate in animal tissues. Even at low levels of intake, it may contribute to cardiovascular disturbances, tissue degradation, etc. in humans. Because of their similar nature, increasing the level of available zinc in soils would probably decrease the uptake of cadmium. Digested sludge contains a low level of cadmium compared to its zinc content (Table 5.2.1). However, the occurrence of cadmium in plants grown on soils treated with sludge should be occasionally determined.

Beryllium usually occurs in soils in only trace amounts, but some containing beryl may have concentrations of 40 PPM. In the acid pH range of soils, beryllium may be highly toxic to plants. The metal is relatively rare and not likely to be found in sludges. It may be found in the waste waters from electronics industries, but most of this metal would probably be lost in the effluent since its chlorides are highly soluble.

Selenium is an essential element for animals, but only trace quantities are required. At a concentration of 0.5 PPM, selenium in milk and water is believed to be harmful for humans. Selenium contents of soils generally range from 0.1 to 2.0 PPM, but there are some soils derived from cretaceous rock that may average 5 PPM selenium. The normal range of selenium in plants is from about 0.02 to 2.0 PPM and, except for accumulator plants, an uptake of 50 PPM or more is toxic. The selenium content of digested sludge is seldom evaluated but should be investigated for land disposal systems.

Vanadium is toxic to man and animals but the toxic concentration has not been established. Most foods contain about 1 PPM. Toxic symptoms occur in most plants at plant tissue concentration of greater than 10 PPM. So far, vanadium has been found to play an essential role only in the nutrition of certain blue-green algae. The concentration of vanadium in digested sludge is also seldom evaluated, but should be investigated if land disposal is considered.

In conclusion, a large variety of chemical elements may be added to the soil with sludge applications. Except for nitrogen, no problems of significance are anticipated. With periodic examination of the soil and plant tissue, loading rates can be adjusted to insure that toxic levels will not be reached.

5.2.2 Historical Record

5.2.2.1 Sewage Sludge

Most of the literature on the agricultural use of sewage sludge reports benefits to soil and crops. Among the plants that have shown a favorable response to sewage sludge when it was supplied in sufficient quantities are wheat, barley, oats, tomatoes, beans, beets, ,berries, potatoes, melon crops, and grass (Morton, 1966) (Allen, 1938) (Baetz, 1942) (Miller, 1966) (Van Kleeck, 1943). In general, adverse effects have been noted only where sludges were concentrated on relatively small areas for several years.

In the early 1940's, experiments were initiated utilizing pot culture and field plots to determine the relative fertilizer value of various sludges (Crowther & Bunting, 1942) (Crowther & Bunting, 1944). It was concluded that sewage sludge was generally equal to farmyard manure and would be superior if supplemented with potash. To a limited extent, sewage sludges could substitute for inorganic nitrogen and phosphorus fertilizers.

Residual benefits from the use of sewage sludge on soils were reported by Lunt (1953) and Merz (1959). These included a 3 to 23 percent increase in field moisture capacity, lower soil bulk densities and an increase in non-capillary porosity with a concomitant increase in infiltration capacity and soil aggregation. Small increases in cation exchange capacity were also noted. As a rule, sandy soils were benefited more by sludge applications than loams.

Only a few investigators cite specific instances of poor plant growth resulting from long term sewage or sludge applications on soils. Trocme, et al. (1951), found that irrigation of soils near Paris with sewage water over several years produced chlorotic conditions in plants which was due to leaching of manganese. Under waterlogged soil conditions, manganese may be reduced to the more soluble divalent form. Also manganese can be displace from organic matter bonds or exchange sites by other metallic cations such as Ca, Ni, Co, Cu, and Zn (Heintze and Mann, 1947). Thus, since the solubility of all heavy metals increase under reducing soil conditions and low pH values and all, except perhaps Pb, would replace Mn on exchange sites, a manganese deficiency may result. However, since all of the heavy metals also compete with Fe, it too may sometimes be deficient for normal plant growth.

Several investigators have reported that sludge additions to soils will eventually cause them to become acid (Hommon, 1931) (Rudolfs, 1928) (Van Kleeck, 1943). Rudolfs (1928) noted that sludge produced acid conditions upon undergoing aerobic decomposition in soil. He found liming was necessary to prevent burning of crops, especially in greenhouse conditions. Van Kleeck (1943) theorized that the breakdown of greases and fats in sludge caused the acidifying effect of sludge. From Table 5.2.1, it may be noted that the hexane soluble materials constitute an appreciable proportion of the total solids in digested sludge, but it is probably that the acid condition is the result of sulfide oxidation and the nitrification process.

Delayed germination was observed by Lunt (1959) when sludge was mixed with the soil prior to planting. Lunt attributed the delay to a high total soluble salt content. Delayed germination of barley was also noted by Merz (1959), who applied liquid digested sludge to a waste soil at rates from 10 to 100 tons of dry solids per acre. He also felt that high soluble salt concentrations in the soil were the cause, since, prior to planting, electrical conductivity values of 5.8 mmhos/cm were found for the soil receiving the 100 tons per acre loading. However, with irrigation and rainfall, the soluble salt concentrations were rapidly reduced to former levels. Lunt (1959) found that his highest rate of sludge application, 260 cu. yds. per acre (50 tons), increased soluble salt concentrations by a factor of 3 as compared to a non-treated soil in a greenhouse experiment.

Considering the electrical conductivity value in Table 5.2.1 for an average digested sludge, it appears reasonable to expect high salt concentrations with continued applications of digested sludge. Salt build-up in soils used for sludge disposal is more likely in arid regions than in areas where normal precipitation would be sufficient to leach some of the soluble salts from the soil horizon. Some recent unpublished results (Miodusjewskie and Hinesly) show that even with 24 inches of digested sludge applied within 70 days on a silt loam soil in Illinois, the electrical conductivity of the soil never increased above 3 mmhos/cm. There was little difference in electrical conductivity values at 7.5 and 38 cm soil depths at the end of the growing season. Thus, in a humid region, it is unlikely that soluble salt concentrations in soils will ever reach those of the applied digested sludge. Natural rainfall will allow necessary leaching. In arid regions excess irrigation water may be required to maintain a favorable salt balance.

Summarizing, if digested sludge is to be applied on a limited acreage requiring heavy applications at frequent intervals of time, special precaution will be needed to maintain a neutral pH, provide drainage, and avoid reducing conditions in the soil. Even with the best soil management practices, intensive use of land for disposal may eventually require additional manganese and iron.

5.2.2.2 Maintenance Dredgings

Sandy sediments have been used for river control and construction works, and to establish or nourish beaches. However, little thought has been given on using the silt, clay, and organic fractions of maintenance dredgings for land reclamation. Since most of the Great Lake's river and harbor dredgings would approximate a silty clay loam soil after sand removal, they would have most of the physical characteristics of an ideal soil. From the standpoint of fertility, it seems reasonable to expect that acid conditions may result as excess water is drained from the dredgings and sulfides are oxidized. This it is probable that liming and fertilization will be required for maintenance dredgings to support a healthy vegetation. However, it is not expected that this requirement would be so great as to seriously detract from their value as a soil material.

5.2.2.3 Fly Ash

As shown in Appendix A, fly ash contains on the average approximately 3.0 percent CaO and 2.0 percent MgO. Thus, fly ash has some low capacity for neutralizing soil acidity. When used in sufficient quantities, fly ash could substitute for agricultural limestone. However, about 30 tons of fly ash would be required to neutralize the same amount of soil acidity as would be accomplished by a one ton application of ordinary crushed limestone. Capp and Engle (1967) applied 600 tons per acre of fly ash on acid strip mine spoil and noted a change from an initial pH of 3 to a pH of 5.1 during the first year. With this change in pH and 1,000 pounds per acre of fertilizer (10-10-10 analysis), they obtained a good growth of Kentucky fescue grass.

The nitrogen content of fly ash is so low that it can be disregarded when assessing the nutrient value of fly ash for disposal on farmland. As a rough average, fly ash contains about 0.13 percent elemental phosphorus, or about twice the phosphorus content of an average soil. However, it is probable that the phosphorus exist in fly ash as tricalcium phosphate $CA_3(PO_4)_2$ and so its availability to plants would be very low. Similarly, unleached fly ash has an average potash (K_2O) content of 2.5 percent, but water soluble potassium is only 0.03 to 0.04 percent of the total. While weathering may result in a fairly constant release, fly ash must also be considered a very poor source of potassium.

Fly ash contains a number of minor elements in such concentrations that toxic levels may be added to soils when applications are of the magnitude needed to effect a soil pH change. The two minor elements boron and manganese appear to be of greatest concern. Holliday, et al. (1958), grew barley on fly ash-soil mixtures containing 0, 12.5, 25, 50, and 100 percent fly ash by weight. The boron content of barley leaves ranged from 15 to 1420 PPM on a dry weight basis, where the lower value was for soil alone and the upper value was for pure fly ash. Barley showed symptoms of boron toxicity when leaves contained 20 PPM of the element, but yields were not reduced until 600 PPM was accumulated. Barley can accumulate 600 PPM boron when it is available in about 200 PPM in the growth media.

Rees and Sidrak (1956) experimented with a variety of crops grown in fly ash and fly ash-soil mixtures. Crops varied greatly in their tolerance to ash and soil mixed in different proportions. With pure ash, Barley and spinach accumulated excessive amounts of Al and Mn in their leaves and exhibited toxicity symptoms peculiar to these elements. Barley also exhibited poor growth, with low levels of ash, but mangold yields were increased by a high proportion of ash. The conclusions were that poor growth of some plants was due to Al and Mn toxicity. However, boron concentrations were not measured, which may have disguised an additional effect. Barley is known to be very sensitive to high soil boron concentrations, whereas mangolds have a high tolerance to the element.

Salter and Williams (1967) incorporated fly ash rates equivalent to 0, 50, 100, and 200 tons per acre in a coarse sandy loam and grew red beets on the plots. All produced satisfactory crop yields, but the 200 tons per acre fly ash application produced a yield 9 percent less than the zero treatment. The reduction was attributed to a boron toxicity. The Tennessee Valley Authority and Ohio Power Co. found that fly ash could be applied at rates up to 200 tons per acre before reduced corn yields and toxic effects were observed (Capp and Engle, 1967).

According to Hodgson and Holliday (1966), fly ash samples from eight power stations had electrical conductivities that ranged from 8.4 to 13.1 mmhos/cm, in comparison to a critical soil conductivity of 40 mmhos/cm. Fresh fly ash samples yielded 1.64 to 3.25% soluble salts. Thus, 200 tons per acre of fly ash may lower yields of crops because of an excessive addition of soluble salts. Although the soluble salts would be readily leached by rainwater to a normal content, the groundwater could be adversely affected. Assuming 25% percolation of a 35"/year rainfall, the 200 tons/acre loading could result in a leachate with a dissolved solids concentration of several thousand mg/l.

In summary, the available information in the literature indicates that fly ash could be applied on farmland at a rate of about 150 tons per acre without causing boron toxicities, except perhaps to very sensitive crops. This rate of application would be equivalent to about 5 tons of limestone. It would supply a small amount of available phosphorus and potassium. Roughly, its value on acid infertile medium texture soils would be about 30 cents per ton. Its value might be considerable more on coarse textured sandy soils. Its highest value would be derived when applied on acid strip mine spoil banks.

5.2.2.4 Water Treatment Plant Sludge

Residues from chemical coagulation operations have not been widely used in agricultural operations. This is understandable as the hydrated aluminum and iron oxides are undesirable to soils. At a soil pH of 6.5 or less, both aluminum and iron can precipitate phosphorus. However, heavy application of aluminum to soils at any pH could produce toxic conditions for most plants.

Where softening is practiced, residues may range up to 90 percent equivalent calcium carbonate. Direct spreading of this sludge on agricultural lands could be highly beneficial, particularly where the soil is acid.

5.3 OTHER ENVIRONMENTAL EFFECTS

5.3.1 Airborne Pollution

Little work has been done on airborne contamination resulting from wastewater disposal practices. Several investigators (Ledbetter, 1964; Glaser and Ledbetter, 1967; Napolitano and Row, 1966, and Randall and Ledbetter, 1966) have indicated that there is airborne contamination with particles containing microorganisms from the aeration tanks of activated sludge units.

There is no reported evidence of airborne pollution resulting from disposal of liquid digested sludge. Investigations into the dispersal of bacteria during irrigation with wastewater show that under properly controlled conditions, and with the sprinkling equipment in use at the present time, there is very little dispersal, except by wind (Reploh and Handlose, 1957). Difficulties with wind dispersal can be avoided by choosing large and remote areas of land and by planting hedges.

Pathogenic fungi have been isolated from wastewater (Dixon and McCabe, 1964) and fungi, not identified as being pathogenic, have been found in sludges (U.S. Public Health Service Report, 1964). However, Dixon and McCabe did not feel that fungi in wastes represented a significant additional health hazard since they are widespread in nature.

5.3.2 Insect Vectors

A number of human diseases are transmitted by flies. Mechanical transmission of pathogens by insects is probably the major cause of concern. Pokrosky and Zina (1938) reported that flies carried helminth eggs on their feet. The fact that flies can and do travel long distances from their breeding sites to human dwellings (Quarterman, et al., 1954; Schoof and Siverly, 1954; Webb and Graham, 1956) indicates that fly problems would not necessarily be located at the sludge spreading area.

Flies typically breed in moist decaying organic material. Studies on fly breeding in wastewater sludges have been reported by Cooke (1954, a, b, c), and Greenberg, et al. (1963). Most reports of liquid sludge disposal on land have not contained any mention of fly problems, or have specifically noted that flies were not a problem (Nusbaum and Cook, 1960, and Anon. ,1968). However, some insect and fly nuisances have been reported in sludge drying beds, sludge lagoons and farmland sprayed with liquid sludge. Spraying with DDT or benzene hexachloride has been helpful in preventing the nuisance (Rawn and Garber, 1950 and Anon. , 1949).

5.3.3 Crops as Vectors

It is not anticipated that crops grown on land receiving digested sludge would necessarily be those intended for human consumption. However, consideration of this extreme case gives a perspective on the significance of crops serving as a mode of disease transmission.

A comprehensive review of the health aspects of wastewater and sludge utilization for crops was given by Rudolfs, Falk and Ragotzkie (1950, 1951). Among the conclusions of their studies were:

a. No evidence has been found that pollutional bacteria, amoeba or helminth eggs penetrate healthy, unbroken surfaces of vegetables or cause internal contamination.

b. Vegetables to be eaten raw can be grown without health hazard in soils that have been subject to sewage irrigation, night soil application, or polluted stream water irrigation in years prior to the season in which vegetables are grown.

c. Vegetables grown under conditions of surface sewage irrigation show no higher coliform concentrations than those grown on normally farmed soil, whether sewage was applied before the plants were set or while the plants were growing.

d. If sewage sludges or night soil are applied on the soil surface, or sewage effluents are applied by overhead irrigation during growth of vegetables, such an application should be stopped at least one month before harvest. If this precaution is taken, the crop will show no higher bacterial contamination than when farm-yard manure or artificial fertilizers are applied.

No case has been found in the literature where crops served as a vector in transmitting disease following land application of liquid digested sludges. However, purveyors of sludge in any form have normally felt obliged to warn against its indiscriminate use. Common state regulations discourage use of sludge (other than possibly heat dried sludge) on soil in which vegetables for raw consumption are grown (Federation of Sewage Works

Association Manual, 1946).

5.3.4 Soil Contact

Survival of bacterial and protozoan pathogens and the persistence of viruses in soils has been considered in previous sections. Of particular concern in considering the significance of human contact with soils receiving wastes, however, is the survival of helminths, or worms. Cram (1943) found that eggs of the large roundworm, Ascaris lumbriocoides, persisted in drying digested sludge for as long as 170 days. Cram also found that hookworm eggs hatched when the sludge dried to a sufficient degree that aerobic conditions prevailed. The larvae developed into the infective stage and stayed alive for up to 62 days during which time the moisture content dropped below 10 percent. These findings confirm what is known from experience in areas where human wastes are indiscriminately applied to land. That is, the helminths can survive for long periods following disposal of wastes on soil and can severely affect the health of individuals in contact with the soil.

However, the possibility of helminth infections as a result of waste applications on land can be circumvented through proper management of the disposal area. Gotaas (1956) noted that,in areas where anerobically digested sewage sludge has been used extensively "as a fertilizer", there appears to be no recorded instances of fecal-borne disease caused by the sludge.

5.3.5 Nuisance Conditions

Successful disposal of sewage wastes has been accomplished for many centuries throughout the world. In particular, peoples of China, India and other ancient countries have long practiced the application of organic material to the land for the purpose of increasing the fertility level. Among European countries, England and Germany have led in the use of wet sludge as a fertilizer supplement. Utilization has been accomplished under many diverse climatic and cropping conditions.

The potential for nuisance conditions from the application of waste products on land,arises mainly from the possibility of odors being produced as a result of decomposition of the organic material. With few exceptions, disposal of liquid digested sludge has generally been accomplished without creating any objectionable effects. An example is the Metropolitan

Sanitary District of Chicago's land reclamation program at the Hanover Sewage Treatment Works. Digested sludge has been utilized for fertilization and irrigation of corn fields located directly adjacent to residential areas. With well stabilized material, no odors are caused (Anon, 1968). In another recent report on liquid application of digested sludge, Lewin (1968) noted that two complaints have been received by the city of Oxford in England regarding odors. Both were from residences bordering sludge disposal areas, and the problem was solved by barring distribution of sludge closer than 100 yd. of inhabitation.

In St. Marys, Pennsylvania, digested sludge has been trucked to farmlands without charge and spread on pastures, meadows and croplands since 1964. One inch truck applications per field area have been made every other year. Land operators and farmers in the area who have utilized the sludge have unanimously expressed approval of its effects. The sludge was judged to promote better growth than barnyard manure and all of the crops and other treated vegetation have responded well. No objections were made of undesirable odor. Farmers in other communities in Pennsylvania utilizing sludge have been equally enthusiastic as those in St. Marys. A common complaint has been that not as much sludge as they desired was available (Evans,1968).

Merz (1959) and Nusbaum and Cook (1960) concluded from experiences with liquid digested sludge application at San Diego that all serious sludge handling and nuisance conditions could be avoided. In describing experiences at Maple Lodge with liquid digested sludge application, Kershaw and Wood (1966) indicated a need for avoiding nuisance conditions along roadways from field runoff during distribution or spillage from trucks, but indicated enthusiastic acceptance of the land application scheme by farmers.

Summarizing this section on other environmental effects, it is conceivable that pathogenic organisms could be dispersed in the air as a result of sludge application by spraying. Likewise, the possibility of transmission by flies or other insects in contact with sludge cannot be dismissed. Crops also are possible vectors; however, by avoiding sludge application to crops intended for human consumption, this problem can be circumvented. Furthermore, experiences with liquid digested sludge application to land indicate that the practice can be carried out without creating odor conditions. A possibility of transmission of diseases, especially helminth diseases, through soil contact exists, but it must be emphasized that no report of disease transmission by any of the mechanisms considered in this section has been found. Nevertheless, vigilance is required since the chance of infection cannot be completely ruled out.

6.0 ENVIRONMENTAL CONSIDERATIONS FOR REGIONAL OCEAN DISPOSAL SYSTEMS

6.1 FACTORS OF CONCERN

Ocean disposal of digested sludge and dredging spoils must be affected to avoid significant degradation of the receiving waters with respect to recreational use or marine life. The design of disposal systems for the western metropolitan areas of San Diego, Los Angeles and Seattle reflects attention to this principle. The results of studies in the nearshore ocean areas potentially affected by the Los Angeles and San Diego outfalls indicate that adequate water quality is being maintained. However, these systems were designed to take advantage of the relatively great depts (200 - 300 feet) of water that occur close to shore along the west coast of the United States.

The beach and nearshore areas (several thousand feet offshore) are used, or potentially may be developed, for recreational bathing along nearly the whole coast of the eastern United States. The economic values of these uses demands that no significant contamination of this nearshore area result from disposal of digested sludge in the ocean. Recreational uses of waters further offshore are primarily based on pleasure boating and sport fishing. These recreational uses require that:

(1) Conditions leading to possible transmission of bacterial, viral or parasitic human diseases must be avoided.

(2) The physical and chemical characteristics of these waters must not be altered significantly.

Ocean disposal of immiscible, floating materials is not acceptable because these materials (greases and oils, for example) do not undergo dilution. The long persistence of such materials leads to a high probability that unacceptable concentrations might be blown into beach areas. The success of the disposal systems of the west coast has been based on careful removal of the floatable materials from digestion tanks and utilizing land disposal for this smaller fraction of the total treated wastes. The injunction by the State of New Jersey against the discharge of ground garbage by New York City is an example of recognition of the particular problems associated with floating materials.

Direct, water-borne transmission of human disease organisms from discharge areas to recreational bathing areas does not seem to be a significant hazard. Public health experience has shown that bacterial standards of 1000 to 3000 coliform per 100 ml have produced no clearly recognizable evidence of high incidence of infection. These concentration levels represent dilutions of 10^3 to 10^4 for discharges of sewage. Since digested sludge may be expected to have a lower coliform concentration than raw sewage, a sludge dilution of 10^4 should also provide adequate bacterial protection.

The ingestion of an infection dose of disease organisms by eating shellfish appears to be a more probable route than chance swallowing of sea water while swimming. Moore (1960) summarized the observations of English bathing experiences as "one may then sum up the evidence on the risk of contracting enteric fever from sea bathing by saying that less than one case a year of those that have come to our notice have given a history that suggested a sea bathing infection, and that the four cases satisfying the criteria set out above were associated with beaches so unsatisfactory that they would be given a high priority for improvement on ordinary aesthetic grounds".

6.2 DILUTION REQUIREMENTS

6.2.1 Transparency

From a recreational point of view, a decrease in the transparency of the coastal waters is undesirable. The general appearance of "dirtiness" will not be accepted. The release of digested sludge solids that may be suspended for long time periods or subject to repeated deposition and resuspension requires adequate dilution to avoid noticeable turbidity. Clear, open ocean waters contain total suspended solids concentrations of 0.1 to 0.5 ppm and coastal waters contain total suspended solids concentrations of 1 to 2 ppm. Although suspended solids concentration and the degree of turbidity are not equivalent, it may be noted that a recommended limit for industrial use of water for beverages is 1 - 2 Jackson turbidity units (McKee and Wolf, 1963). Therefore a reasonable permissible increase in suspended solids might be taken as 1 ppm without a clearly visible change in transparency. Thus, digested sludge with an approximate initial concentration of 3.5 percent will have to be diluted approximately 3×10^4 times in order to preserve recreational use of the nearshore waters.

It must be recognized that this criterion is arbitrary and is a judgement factor based on other experience. In order to fully evaluate transparency limitations further work is required to establish the effect of the many variables involved, such as depth of water and type of sludge.

6.2.2 Nutrients

A second consideration in avoiding degradation of recreational use is the nutrient content of the digested sludge. The development of large populations of algae may seriously damage the coastal waters. The "red tide" blooms of Gymnodinium brevis along the west coast of Florida are a notorious example. During the summer of 1968, a number of beaches in northern New Jersey were closed by public officials because of a "bloom" of algae that strongly discolored the nearshore waters for a week.

Both the quantity and chemical form of constituents in the sludge have a bearing on the possible stimulation of algal growth. Information on algal nutrition is inadequate to allow a precise estimate of the probable effects of sludge discharge. For example, recent unpublished work at The Johns Hopkins University on a few species of marine algae shows a greater growth per unit of nitrogen when the nitrogen is supplied as amino acids rather than as ammonium or nitrate salts. Changes in he dominant species might result from nutrient enrichment with digested sludge, and these species might be either desirable or undesirable.

Phosphorus and nitrogen have been frequently identified as a growth limiting constituents in coastal waters and are considered here to illustrate the effect of nutrient concentrations on the design of sludge disposal systems. The concentration of phosphorus as inorganic phosphate in the shelf water off New Jersey has winter time maximum values of 0.03 ppm or less. The nitrate-nitrogen concentrations have maximum values of 0.08 ppm or less (Ketchum, Vaccaro and Corwin, 1958). The more fertile ocean waters, such as those upwelled off the Peru coast, have phosphate concentration of 0.1 ppm and nitrate-nitrogen concentrations of 0.6 ppm. It seems undesirable to increase the concentrations of phosphorus and nitrogen to levels above those observed in the fertile waters and a reasonable permissable increase is taken as 0.05 ppm phosphorus and 0.5 ppm nitrogen for the purposes of this report.

Both liquid and sludge solids must be considered as a source of nitrogen and phosphorus. Although the composition of digested sludges may vary, for a 3.5% sludge, the total phosphorus and nitrogen content will be taken as 800 ppm and 1500 ppm respectively. A considerable portion of the organic forms of these elements may be mineralized in the receiving waters (Gunnerson, 1961). Approximate values for available nutrients are assumed to be 500 ppm for phosphorus and 1000 ppm for nitrogen. The required dilution to achieve acceptable concentration levels is thus estimated to be 10^4 for phosphorus and 2×10^3 for nitrogen. These dilutions appear to be the minimum that can be tolerated to avoid unsightly algal growth.

The effects of digested sludge on marine life may depend partly on the transparency and nutrient considerations that influence recreational uses as outlined previously. The decrease in transparency would reduce the depth to which photosynthesis could occur and, thus, has a potential effect on primary productivity. Doubling the suspended solids concentrations would roughly double the extinction coefficient and the depth at which photosynthesis is balanced by respiration would be reduced by a factor of two. It is not possible to anticipate the effect of reduced transparency on the total primary productivity because of possible adaptation by the algae and possible changes in dominant species with the altered distribution of light with depth.

The transparency effects would be partially offset by the increased nutrient concentrations derived from the digested sludge suspension. Our knowledge of the nutrition of marine algae is entirely inadequate to permit a quantitative evaluation of the probable effects of added nutrients. However, observations in various parts of the ocean have shown a rough correlation between nutrient concentrations and standing stock of algae. Dilution of 3.5 percent digested sludge suspensions by approximately 10^4 would produce nutrient levels comparable to those found in the most fertile ocean waters, as described above, and increased production of algae must be expected.

6.2.3 Dissolved Oxygen Concentration

The dissolved oxygen concentrations in the coastal waters are important considerations in designing the disposal systems for digested sludge. As noted above, dilution of 10^4 is desirable for nutrient concentration control (assuming available phosphorus content of 500 ppm) and dilution of 3×10^4 is desirable for turbidity control. Since improved treatment to reduce the discharge of phosphorus with sewage treatment plant effluent may be widely used in the future, it seems probable that the phosphorus concentration in the sludge will increase. Incorporation of all phosphorus in the sludge would increase the phosphorus concentration in the sludge by approximately a factor of three, based on assumed values of 0.002 pounds phosphorus per capita per day in present sludge and 0.004 pounds phosphorus per capita per day in present liquid effluents. Since both the turbidity consideration and the probable future phosphorus consideration require dilutions of 3×10^4, this value is taken as a design criterion for considering dissolved oxygen changes.

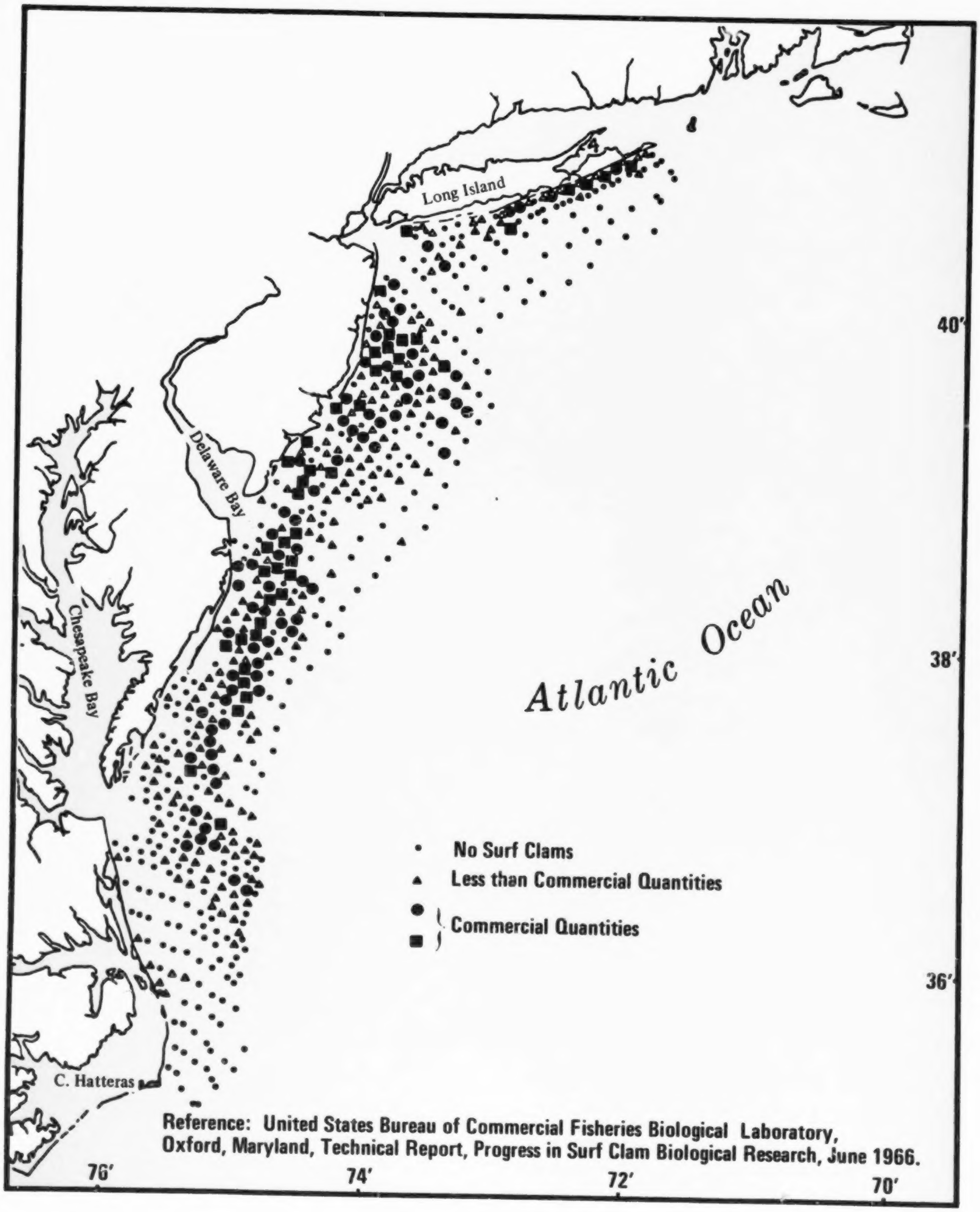
Dilution of 3×10^4 for a 3.5 percent sludge would increase suspended solids in the receiving waters by 1.2 mg per liter. If the organic material in the solids constitutes roughly 40 percent of the total, the suspended organic addition would be 0.5 mg per liter. Reliable information on the oxygen demand of dilutions of digested sludge with sea water is not available, but a conservative assumption may be derived as follows. The 5 day, 20C BOD of the suspended solids in domestic sewage is reported to be 140 mg per liter when the organic matter concentration is 215 mg per liter (Fair and Geyer, 1954), or the 5 day, 20C BOD is roughly two-thirds percent of the organic matter concentration. The ratio of sludge BOD to organic content will be less than this value since the more readily decomposable materials will have been stabilized in the digestion process. However, let us assume that approximately one-half of this BOD is exerted in the first 48 hours following discharge. Furthermore, let us say that the dilution criterion of 3×10^4 will be met within a few hours after discharge and that further dilution would not occur until after 48 hours. Then the oxygen demand of sludge can be estimated as being one-third of the organic matter concentration or 0.17 mg per liter. The conclusion is that the direct oxygen demand based on BOD does not appear to be a serious problem with the 3×10^4 dilution criterion.

However, the discharge of digested sludge will affect the dissolved oxygen distribution in coastal waters by a more involved process that just ordinary BOD. As noted, the dilution criterion of 3×10^4 was based on phosphorus concentration increases to levels comparable to those observed in the more fertile oceanic areas. One characteristic of these fertile areas is subsurface depletion of oxygen resulting in a layer of low oxygen concentration. As pointed out by Redfield (1958), there is a correlation between the phosphate concentrations in the North Atlantic and North Pacific waters and the oxygen concentration in the oxygen minimum layer. Redfield also discusses the general covariance of phosphorus, nitrogen, organic carbon and oxygen during the growth and decomposition of marine planktonic algae. The observed correlation is that 1 atom of phosphorus corresponds to 250 atoms of oxygen and, thus, the utilization of 0.1 ppm of phosphorus during algal growth would produce 11.6 ppm of oxygen and, conversely, the complete decomposition of the algal cells to release the 0.1 ppm of phosphorus would be associated with the consumption of 11.6 ppm of oxygen. The solubility of oxygen in coastal water at the mean temperature of 12° C is 8.4 ppm, so the possible algal production under the 3×10^4 dilution criterion would lead to complete consumption of the oxygen in the late spring and summertime when the coastal waters are thermally stratified. This result assumes complete decomposition of the algae and complete loss of the oxygen produced. A more probable condition is partial decomposition of the algae and retention of perhaps 10 to 20 percent of the oxygen produced. Then the deeper waters would not become anoxic but low levels (1-3 ppm) of dissolved oxygen would occur.

Fertilization of the continental shelf waters would increase the primary productivity and potentially increase the harvest of fish that derive their food from zooplankton that feed on the algae. However, the concomitant reduction of oxygen in deeper waters would adversely affect bottom-feeding fish and shallfish. As shown in Figure 6.2.1, the distribution of commercial levels of surf clams is widespread. There also exists a large population of mahoganey clams in the area of deeper water further offshore, which are not commercially harvested now but are a potential food supply. The oxygen concentration changes postulated above could prove detrimental to this natural resource.

FIGURE 6.2.1

SURF CLAM SURVEY CRUISES


Long Island
Delaware Bay
Chesapeake Bay
C. Hatteras
Atlantic Ocean
• No Surf Clams
▲ Less than Commercial Quantities
● ■ Commercial Quantities
40'
38'
36'
76'
74'
72'
70'

Reference: United States Bureau of Commercial Fisheries Biological Laboratory, Oxford, Maryland, Technical Report, Progress in Surf Clam Biological Research, June 1966.

6.3 SHELLFISH CONTAMINATION

6.3.1 Viruses

Another potential effect of sludges, discharged to the ocean, is the contamination of shellfish with viruses. Infectious hepatitis, in particular, has been identified as transmissible through shellfish that are eaten raw. The recent paper by Metcalf and Stiles (1968) describes the isolation of other enteric virus pathogens from oysters growing in estuarine waters that met the standard of not over 70 coliform median MPN per 100 ml. These results raise serious questions concerning the effectiveness of bacterial standards in controlling viral problems. However, none of the shellfish that are harvested from the shelf and coastal waters are eaten raw. The surf clam is cleaned and the portions most likely to be contaminated are discarded. Other clams are not cleaned and the whole organism is eaten. The cooking of these clams tends to reduce the hazard of viral contamination, but there is no evidence on the effects of different kinds of cooking, in particular, the rapid frying of breaded clams with short heat exposures. Furthermore, it is not possible to demonstrate the level of the hazard since the virus of infectious hepatitis is not identifiable in the sludge or shellfish with present techniques. One possible resolution of this problem, if it arose, would be regulation of the cooking procedures used to process the clams.

6.3.2 Heavy Metals

Another possible effect of sludge discharge to the ocean arises from the heavy metal content. Although standards for trace metals in shellfish have not been established, some of the medical aspects associated with zinc, copper, cadmium, lead and mercury have been reviewed by Pringle and Shuster (1967). They suggested an interim guideline of 1500 ppm Zn, 100 ppm Cu and 2 ppm for Cd, Pb, Mg and Cr as wet tissue weight in shellfish. Digested sludge analyses from communities having extensive industrial activities have reported concentrations of lead and chromium up to several hundred ppm. With sludges containing 100 ppm of those elements and assuming a shellfish guideline concentration of 2 ppm, extensive dilution of the sludge will be necessary to avoid metal contamination of shellfish. Analyses of quahogs and clams summarized by Pringle and Shuster show concentrations of 0.5 to 2 ppm for Cd, Pb and Cr. The sea water concentrations of these metals are 0.1 to 1 ppb, or the concentration factor for the metals between sea water and shellfish is roughly 5×10^3. With a sludge metal concentration of 100 ppm, the required dilution would be 2.5×10^5 to avoid contamination of shellfish to levels above the 2 ppm wet weight guideline. Since the rate of release of heavy metals from shellfish is very slow, it would not be practical to attempt purging once contamination has occurred. Therefore it may be necessary to devise special methods of disposal for these sludges.

6.4 CONCLUSIONS

To avoid degradation of the shelf waters, altered transparency, nutrient concentrations and dissolved oxygen concentrations have been singled out as controlling factors. A dilution of 3×10^4 was indicated as a possible acceptable design criterion for these factors. Even then sludge concentrations of cadmium, lead, chromium and mercury of more than 10 ppm are not compatible with such a criterion and such sludges must be considered as cases requiring special disposal schemes.

However, this is a very large dilution demand. The volume of fresh water added to the continental shelf averages 5×10^{12} cubic feet per year (Ketchum and Keen, 1955). With an average freshwater inventory of six percent of the total shelf volume, the renewal rate, or total exchange of shelf water with the deeper ocean regions, must be roughly 8×10^{13} cubic feet per year. For the year 2000 an estimated 4.5×10^9 pounds (dry solids) of digested plus chemical sludge per year is projected for the Atlantic megalopolis area. In terms of total phosphorous and nitrogen, 10^8 lbs and 2×10^8 lbs respectively would be included. At a concentration of 3.5 percent, this material would be added at a rate of 2.1×10^9 cubic feet per year. For 3×10^4 dilution, the required "new" water would be 6.3×10^{13} cubic feet per year. Therefore, the total sludge production in the year 2000 could require most of the total renewal of shelf water.

Furthermore, this over-simplified computation is optimistic. The distribution of the fresh water shows a gradient of two across the shelf. Since the sludge would be added at roughly the same localities as the fresh water (mouths of estuaries), a similar gradient in sludge constituents would be expected. Therefore, it may not be possible to meet the 3×10^4 criterion in the nearshore area. This suggests that ocean disposal on the shelf could be tolerated for a while longer, but may not be an acceptable long range solution for all communities on the Atlantic Coast. Additional data on this subject will soon be forthcoming from other studies, such as the current Sandy Hook Marine Laboratory investigation on "The Effects of Offshore Waste Disposal Practices in the Marine Community".

7.0 TREATMENT OF WASTE MATERIALS

7.1 EVALUATION OF NEED

In the preceeding sections, the effects of land and ocean disposal of wastes on the environment have been considered. In cases in which waste constituents may have potentially adverse effects, several alternatives are available. One possibility is to reduce the waste application rate per unit area or volume of the environment to reduce the concentration of the offending constituent. Another is to exercise waste management practices, to alleviate the potential problem. Finally, treatment to remove or alter the troublesome waste constituent could be provided at some time prior to the point that control of the substance in the environment is lost.

On the basis of the discussions in Section 5, treatment for two sludge constituents, communicable disease agents and nitrogen, may be warranted for land disposal. Treatment for elimination of communicable disease agents may be justified to improve public acceptance of regional waste management practices. Nitrogen removal could be warranted because it would permit higher rates of waste application per unit area of land. The rationale for considering treatment for these two constituents and possible treatment methods are included in the sections which follow.

7.2 TREATMENT FOR COMMUNICABLE DISEASE AGENTS

The diseases of most concern are the enteric diseases of man. The infectious agents for these diseases are found in sewage from infected individuals and can be contained in fresh sewage sludge. However, the environment outside of the human body in generally hostile to these agents of disease. Appreciable reductions in the number of pathogenic organisms and viruses occur during anaerobic digestion of sludges. Indeed, viruses have not been isolated from digested sludge. When wastes are discharged to the land, further reductions in their numbers occur with time. In addition, the disease causing agents are selectively removed as water passes through soil.

Hence, the danger of transmission of disease through disposal systems is considered minimal. No report has been found of disease transmission as a result of land application of liquid digested sludge. This includes disease transmission from the disposal area by the water route, airborne pathogens, soil contact, crops, or insect vectors.

Yet, the response to the question "Is it conceivable that disease could be transmitted from well managed regional waste disposal systems?" would have to be "yes". The public is understandably concerned that wastes - particularly their neighbors' - be managed in a fashion which eliminates the possibility of disease transmission. Thus, it may be necessary to provide additional safeguards against disease transmission to secure the public acceptance required for the success of regional waste management systems. Possible techniques for such treatment - if needed - are considered here.

7.2.1 Storage

A likely component of land disposal systems would be storage facilities to contain wastes until they can be applied to land. Similar to an anaerobic digester or the soil, the environment of a sludge lagoon would be hostile to pathogens. For example, the results of McKinney, et al. (1958) show about 50 percent additional reduction in *Salmonella typhosa* densities when sludge digested for 6 days was maintained under anaerobic conditions for an additional two weeks. Therefore, a further reduction in the number of pathogenic organisms and viruses would be expected.

Unfortunately, this treatment technique suffers the same shortcomings as the other processes which reduce the numbers of pathogens and virus. That is, assurance could not be given that this technique completely eliminates the possibility of disease transmission. Hence, while storage of wastes prior to disposal would reduce the possibility of disease transmission, it could not satisfy possible public demands for a "fail safe" system of waste management.

7.2.2 Heat Treatment

Heat treatment of material is a common method of destroying pathogenic organisms and is widely used in food canning and milk industries. Complete sterilization (destruction of all agents) can be readily accomplished by treatment at extreme conditions of temperature. However, in certain cases, less severe treatment serves to destroy organisms of greatest concern at a much lower cost. Such treatment is referred to as pasteurization.

Scant information exists concerning the application of heat treatment to digested sewage sludge. In the application of dewatered sludge to land, only heat-dried sludge has been considered to be free of pathogenic agents (Andersen, 1959). However, food industry practice, particularly milk pasteurization, can provide some guidelines for the selection of temperature-time criteria for the pasteurization of digested sludge.

The prime consideration for milk pasteurization is the need for killing the organism responsible for tuberculosis. The resistance of this organism is commonly considered to be greater than other non-sporulation organisms (Birkeland, 1949). Two methods are used to accomplish this pasteurization; the first is a high temperature (72° C), short time (15 seconds) process, and the second is a low temperature (62° C), long time (30 minutes) process. A secondary consideration in selection of these time-temperature parameters is the need to avoid a "cooked" taste in the milk.

Kaplan and Melnick (1954) reported that human coxackie virus in water, milk and cream, were destroyed by either of the above methods. With regard to the bacteria in digested sludge, Davis (1948) has indicated that the vegatative cells of all pathogenic bacteria, as well as fungi and viruses, are sterilized within a few minutes at 50° to 70° C. In the absence of experimental evaluation of the required time-temperature conditions for sludge pasteurization, two limiting cases have been chosen to illustrate the economics of such a treatment process, vis. 70° C for 30 minutes, and 76° C for 2 minutes. These conditions are somewhat more severe than conventional milk pasteurization practice to compensate for agglomeration of particles and the specific heat transfer characteristics of sludge. A recent report of sludge pasteurization in Germany (Vater, 1969) indicates some agreement with the long time-low temperature conditions listed here.

The method chosen for heat treating digested sludge is a submerged combustion system similar to that used in Philadelphia for sludge heating prior to digester loading (Furczyk, 1955). In this process, combustion products are fired directly into the liquid, giving immediate heat transfer. This direct heating system avoids the problem of fouling of heat transfer surfaces which would be encountered in an indirect heating system, such as the conventional shell and tube heat exchanger.

Figure 7.2.1 shows the cost (dollars per ton of dry solids) as a function of capacity for the treatment conditions outlined previously. Costs are indicated for sludge concentrations of 3.5 and 5 percent solids by weight. In all cases, it is assumed that sludge is available from the digesters at 90° F. Capital is charged at 10 percent per annum at 1969 equipment costs.

7.2.3 Chemical Disinfection

Tyler, et al. (1950), reported that there was "an almost complete absence of data on . . disinfection of sludges . . ". Little additional work in this area has been accomplished since 1950. However, there has been significant progress in the field of disinfection of water with respect to the effect of various chemicals on pathogenic bacteria, virus and protozoa and, within limits, this work can be applied to disinfection of sludges.

7.2.3.1 Factors

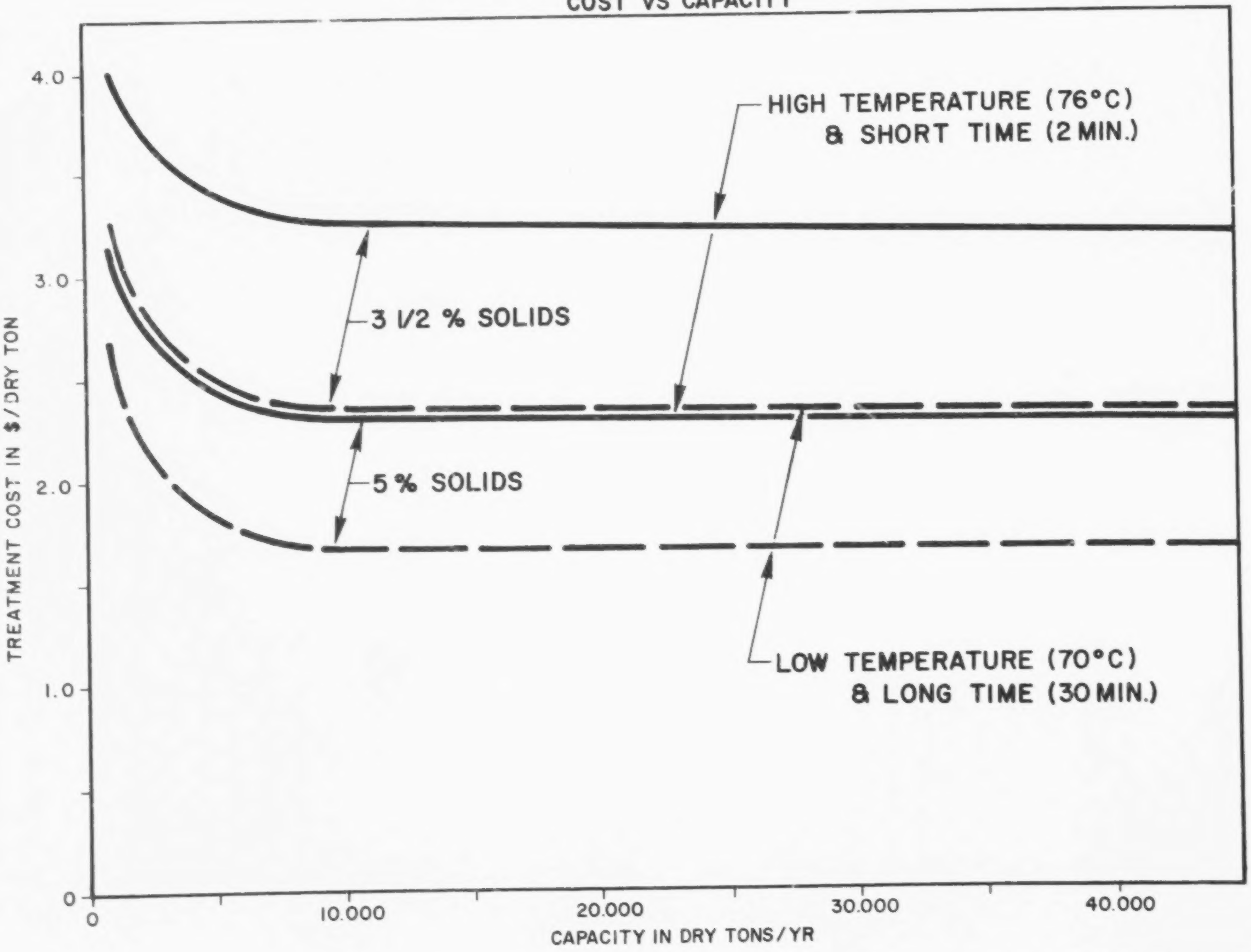
Some of the conditions which govern disinfection with chemicals are considered briefly in the sections which follow. Because disinfection techniques in this country have developed principally from experience with chlorination, many of the basic chemical disinfection concepts relate specifically to that practice.

Resistance of Organisms

As with disinfection of water, destruction of pathogenic bacteria, protozoa and viruses contained in human waste discharges would be the goal of sludge disinfection. Although attention has commonly been given principally to the pathogenic bacteria, the cysts of the protozoa Endamoeba histolytica and the more important viruses are probably more resistant to most forms of chemical disinfection. Morris (1966) has noted that the chlorine dose required to kill the infectious hepatitis and polio viruses is about 20 times that required to kill enteric bacteria. It should be noted that, in spite of this evidence, the

FIGURE 7.2.1

HEAT TREATMENT OF DIGESTED SLUDGE
COST VS CAPACITY



coliform group is ordinarily used as the indicator organism in disinfection studies because of the extreme difficulty of assaying for human viruses.

Composition of Material

The influence of sludge characteristics is of special importance in considering possible chemical disinfection of sewage sludge. Suspended solids in the sludge may harbor organisms and protect them from the disinfectant. Other substances in sludge may react with disinfectants to render them less effective or completely ineffective. For example, disinfectants which are strong oxidizing agents would react with organic material and other reduced substances in the sludge such as sulfides. The disinfectant would be reduced and rendered innocuous in the reaction. Because sludge contains extremely high concentrations of reduced substances, this "demand" for the disinfectant for unproductive purposes would be significant. The disinfectant which remains after satisfaction of the demand is termed the "residual".

Disinfectants can also react with constituents in the sludge with the result that their disinfecting capabilities are attenuated but not destroyed. For example, reaction of chlorine with ammonia (which is in high concentration in digested sludge) produced chloramines which are less effective disinfectants. Similarly, organic chloramines would be produced on chlorination of sludges.

Temperature

Chemical disinfection takes place more rapidly at higher temperatures. In disinfecting digested sludge, an advantage probably would be realized in treating the sludge immediately after it was removed from the warm digester.

pH

The intensity of hydrogen ions in the material being disinfected influences both the organisms and the nature of the disinfecting chemical. Chlorination offers the most notable example of this effect. Active chlorine which is not combined with ammonia is called free chlorine. It can exist either as hypochlorous acid or as the hypochlorite ion. The undissociated acid is a much better disinfectant than the hypochlorite ion. The relative abundance of the two forms of chlorine is highly pH dependent with the less effective hypochlorite ion predominating at high pH values. Hence chlorination is most effectively carried out at low pH.

Time

Time is required for disinfectants to accomplish their task. In water treatment practice, contact times in the order of 30 minutes are common. In general, a longer contact time may be substituted for a higher disinfectant concentration.

Contact Opportunity

Disinfection facilities must be designed to assure uniform intimate contact of disinfectant and microorganisms and to avoid short-circuiting of flow through the disinfection facilities. It is also necessary to assure that opportunity is afforded for contact of disinfectant with organisms shielded by sludge solids. Disintegration of solids prior to disinfection might be warranted with sludges.

7.2.3.2 Chemical Disinfectants

Although it is possible to accomplish disinfection with a variety of chemicals, chlorine has traditionally been a prime choice. As Morris (1966) has pointed out, while other chemicals may have a specific property which is better than that of chlorine, none have a combination of properties which is more favorable. However, because of differences between water and sludge, it is appropriate to reevaluate the choice of disinfectants. Some of the chemicals which might be considered for sludge disinfection are discussed below.

Metal Ions

Organic material in sludge would not exert a demand for metal ions. However, they probably would be precipitated and rendered ineffective by sulfides in the sludge and this "demand" would have to be satisfied. Of the metal ions, silver probably comes the closest to having the desired disinfecting capabilities. However, Faber (1961) has indicated that silver was not a satisfactory disinfectant for sewage because its action is inhibited by colloidal and suspended organic solids. Also, its concentration in drinking water is limited to 50 ppb by the U. S. Public Drinking Water Standards (1962). Chambers (1961) has suggested that application of silver is limited to situations where cost is not a primary consideration. Hence, the metal ions do not appear to offer attractive possibilities for sludge disinfection.

Ozone

Ozone is a strong oxidizing agent which is widely used in Europe for disinfection of public water supplies. However, Stumm (1958) has noted that the strong oxidizing characteristics of ozone would lead to an exorbitant demand for the chemical in sewage disinfection. This disadvantage would be more pronounced with sludge disinfection.

Halogens

Chlorine, iodine, and bromine have all received attention as disinfectants. Chlorine in proper doses should be satisfactory for sludges. One disadvantage, previously mentioned, is its reactivity with sludge constituents.

Iodine has been used with success in disinfection of water supplies (Black, *et al.*, 1965). It has the advantage of not being as reactive with organics as chlorine and does not form less reactive iodamines. The effect of iodine in water percolating through or running off of land would need to be considered because of the physiological implications of iodine with regard to thyroid conditions.

Bromine is even more reactive with organic matter than chlorine. Therefore, a larger dosage of bromine would be required for disinfection. Also, Wyss and Stockton (1947) have noted that its effectiveness becomes limited above pH 5 due to formation of the less effective hypobromous acid and hypobromite ion.

Of considerable importance in comparing the applicability of halogens is their relative cost. Chambers (1961) indicated that in 1961 bromine cost about 3.5 times as much as an equivalent amount of chlorine, and iodine cost about 18 times as much as an equivalent amount of chlorine.

Adverse pH Conditions

Studies concerned with water disinfection have shown that a high pH will result in the destruction of certain microorganisms. Around 1910, Dr. A. C. Houston of the Metropolitan Water Board, London, England, first proposed excess lime treatment as a method for disinfection of public water supplies. He concluded that 200 mg/l of lime would disinfect Thames River water in 5 to 24 hours. Later examinations of lime softened and filtered waters at six water softening plants in Ohio confirmed that E. coli was substantially reduced by this treatment (Hoover, 1924). The disinfection was found to proceed when pH values were in excess of 9.5. Destruction of E. coli at the Columbus Ohio plant was complete in 5 hours when approximately 50 mg/l of lime was applied in excess of the amount required for softening (Hoover, 1962).

In 1943, Wattie and Chambers tested the effect of high pH on *Aerobacter aerogenes*, *Pseudomonas pyocyaneous*, *Eberthella typhosa*, *Shigella dysenteriae*, and four strains of *Escherichia coli*. Death rates of all organisms increased with increasing pH over the range from 9.0 to 11.5. In almost all cases, the kill was nearly complete within 5 hours at pH 11.0 at room temperature. Similar results were reported as a result of a study conducted by Riehl, *et al.* in 1952.

A question remains, however, concerning the destruction of the more resistant microbiological forms such as virus at high pH. A search of the literature revealed few reports on which to base even preliminary judgements. However, some data on the inactivation of enteroviruses by high pH has been accumulated by Berg, Chang, and Berman in a report on military disinfectants to the U. S. Army Natick Laboratories. This report has been referenced in the volume edited by Berg in 1965 on the "Transmission of Viruses by the Water Route". It is implied that a high pH is effective in inactivating virus.

All of the work above was concerned with the disinfection of water. Although it is often hazardous to extrapolate results, these studies indicate that it may be possible to destroy certain pathogenic organisms in digested sludge with pH values of 11.0 or higher. Prolonged detention times, perhaps on the order of a day, might be required. Heating of the sludge might increase the death rate.

The lime dosage required to reach a pH of 11.0 will vary with the sludge composition. If the alkalinity of a digested sludge were 3000 mg/l $CaCO_3$ equivalent, the lime requirement might be around 1710 mg/l as CaO. This would include about 1680 mg/l of lime to convert bicarbonate alkalinity to carbonate and about 30 mg/l of lime to provide hydroxyl ion.

7.2.3.3 Experience with Sludge Disinfection

Disinfection of sludges is not ordinarily practiced. Furthermore, only limited information from laboratory scale studies is available.

Connell, et al. (1958), investigated disinfection of undigested primary sludge. They found that they could reduce coliform densities to less than 100 per 100 ml by limiting solids concentrations to less than 1 percent by dilution, homogenizing test samples in a Waring blender, and dosing with sufficient chlorine, bromine or iodine to give a halogen residual of at least 10 mg/l of equivalent chlorine after two hours in a stirred contact vessel. Over the solids concentration range from 1 to 1.5 percent, the required chlorine dosage to provide the desired residual ranged from about 450 to 880 mg/l and the dosage per unit of solids averaged 4.6 g of chlorine per 100 g of solids. However, the required dosage per unit of solids depended on the solids concentration. For example, the average chlorine dosage was 3.2 g/100 g solids, or 140 mg/l when sludge with a concentration of 4400 mg/l was used, and 5.7 g/100 g or 880 mg/l when the suspended solids concentration was 15,400 mg/l. In addition to the higher chlorine demand of the more concentrated samples, higher chlorine residuals were required. While it is difficult to extrapolate these data to more concentrated sludges, if 5 percent sludge were diluted to the concentration range studied by Connell, et al., the total chlorine requirement would amount to about 1600 to 2850 mg/l on the basis of the concentrated sludge volume.

Connell, et al., also found that chemically equivalent residual concentrations of chlorine and iodine had equivalent colicidal effects while lower bromine residuals would be adequate. The required dose of bromine was about twice that required for chlorine, while the required iodine dose was two-thirds of the chlorine dose. Combining these relative dosages with relative costs cited previously would indicate that to accomplish the same degree of disinfection by use of bromine would cost about 7 times as much as chlorine and use of iodine would cost 12 times as much.

Chlorination of both raw and digested sludges was investigated by Tyler, et al. (1950). Digested sludge was found to require a higher dosage per unit volume than raw sludge for an equivalent amount of disinfection. The chlorine demand of digested sludge was several times that of raw sludge and varied directly with volatile solids concentration. However, the chlorine requirement per capita would be about the same regardless of the type of sludge used, because of the volume reduction accomplished in anaerobic digestion.

As with Connell, et al., Tyler and his co-workers worked with diluted and homogenized samples. Normally the sludge was diluted several hundred times and most of their data were for concentrations less than 1000 mg/l (0.1 percent). Samples were homogenized in a laboratory blender and the necessity of chlorinating immediately after homogenization before reagglomeration could occur was noted. Contrary to the findings of Connell and his co-workers, the chlorine requirements per unit of solids increased with greater dilutions.

The data of Tyler, et al., indicate that the required chlorine dose for 99 percent kill of E. coli in diluted samples of digested sludge ranged from about 17 mg/l at 300 mg/l of total solids to about 33 mg/l at 1400 mg/l. It is not possible to extrapolate these values to concentrated digested sludge. However, if a sludge with 5 percent solids were diluted to 1000 mg/l, the required chlorine dose from Tyler's data would be about 1350/mg/l based on the volume of the original undiluted sludge. For raw sludge, the dose under the same circumstances would be about 1100 mg/l.

Tyler, et al., suggested that it might prove advantageous to chlorinate the entire raw sewage flow rather than to attempt sludge chlorination. Dilution of sludge would then not be required. Because of the current trend toward chlorination of waste treatment plant effluents, the suggestion is more appropriate in 1969 than when it was made in 1950. Adequate disinfection of raw sewage would eliminate the need for disinfecting either sludge or the final effluent. Also, any objection to discharge of water containing a residual of chlorine to the natural environment would be avoided. The estimated doses of chlorine for disinfection of raw sewage listed in the Water Pollution Control Federation's Manual of Practice (1951) is 6 to 12 mg/l for fresh sewage and 12-25 mg/l for septic sewage. The same source estimates the required dosage for effluent from an activated sludge plant to be 2 to 8 mg/l. Taking the midpoint of these ranges and considering the waste to be "semi-fresh", the additional chlorine requirement for disinfecting raw waste is about 9 mg/l. The cost of the difference could be taken as the price of sludge disinfection. Considering the waste sludge volume to be 0.3 percent of the total flow, the equivalent dosage would be 3000 mg/l of sludge.

7.2.3.4 Conclusions

The literature offers little information about sludge disinfection. It could be expected, however, that solids, organic material, and nitrogenous compounds would complicate sludge disinfection by interferring with the added chemicals. A number of chemicals might be utilized, but experience to date has been limited to the halogens. In proper concentrations, all are capable of disinfecting sludges. Considering the factors influencing the required dose and the relative costs of the halogens, the use of chlorine is recommended. However, a major drawback of this method of disinfection is the tremendous increase in chlorides. This would be particularly disadvantageous for utilization of sludge in agricultural operations.

In previous work with sludge disinfection, homogenization and dilution have been provided to assure adequate contact of disinfectant and infective agents. Hence, sludge disinfection facilities might require comminution, dilution, disinfection and concentration facilities in that order. It is estimated that the required chlorine dose using this procedure would be in the order of 2500 mg/l based on the original undiluted volume of a 5% suspension. This is equivalent to adding 100 lbs. of chlorine per ton of dry solids. With chlorine at $60 per ton, the chemical cost for chlorine disinfection would amount to $3.00/ton of dry solids.

Another alternative with chlorine would be to disinfect the entire raw waste flow. This method would be particularly attractive in cases where chlorination of final effluent was to be provided anyway. The additional chlorine required to disinfect raw rather than treated waste could be taken at about 9 mg/l, which is equivalent to approximately 3000 mg/l on the basis of the waste sludge volume.

One additional method for disinfection that might be promising is addition of lime to obtain a high pH. Assuming a 5% sludge with an alkalinity of 3000 mg/l $CaCO_3$ equivalent, the lime requirement would be in the order of 1710 mg/l as CaO. This is equivalent to adding about 70 lbs. of lime per ton of dry solids. With lime at $20/ton, the chemical cost for disinfection would amount to $7.00 per ton of dry solids. However, further investigation is necessary to ascertain if digested sludge will be effectively disinfected under this assumed pH condition.

7.2.4 Irradiation and Other Physical Methods

Irradiation

Some of the most definitive work on irradiation of wastes has been performed recently under contracts to the Metropolitan Sanitary District of Chicago by Battelle-Northwest and Purdue University. Their results indicate that disinfection of sewage sludges may be achieved as well as degradation of certain refractory compounds. An increase in settling and filtration rates of waste activated sludge was also noted, but little improvement in the settling characteristics of digested sludges was observed.

Both Battelle and Purdue found that organism kill was a function of total dose and not dose rate. Complete coliform kills required radiation levels of approximately 200,000 RADS. Based on the limited work to date, it appears that the cost of sludge irradiation is in the same order of magnitude as complete dewatering and incineration. Therefore, unless other significant benefits were also achieved, such as possibly as stabilizing sludge without digestion, use of irradiation for sludge disinfection appears economically prohibitive.

Other Physical Methods

Disinfection is possible by physical techniques such as exposure to ultra-violet light, ultrasonic vibrations, or high pressures. In general, these techniques have not been widely and successfully applied in large scale applications.

Recently, interest has been shown in electrohydraulic sterilization (Allen and Soike, 1967). In this process, high voltage electricity stored in capacitors is discharged across a gap in the liquid being treated. Bactericidal action is produced by the 2,500 to 5,000 atm pressure pulses generated or, more probably (Gilliland and Speck, 1967) by oxidation reactions mediated by free radicals produced by the electrohydraulic shock. Sterilization of raw sewage by this technique has required 5 watt hr/gal (Allen and Soike, 1966). No application of the technique to sludge is known.

7.3 TREATMENT FOR NITROGEN REMOVAL

In Section 5, it was noted that high rates of sludge disposal to land might result in the application of more nitrogen than would be removed by crops and gas formation. The result could be that nitrogen would be biologically oxidized to the mobile nitrate form and carried to ground or surface waters. High concentrations of nitrates in water are of concern because they may lead to illness in man and animals and also because they may serve as a nutrient for algal growths in surface waters.

Hence, nitrogen appears to be a principal limiting factor for land disposal. Considering that other possible constraints on the rate of sludge application may not be operative until appreciably higher application rates are reached, removal of nitrogen may extend the capability of land disposal systems. By permitting greater amounts of digested sludge to be applied to land, it would reduce distribution and management costs. Therefore, optimization of the overall regional waste management system requires consideration of possibilities for nitrogen removal.

7.3.1 Nitrogen Removal Experience

No research or experience directly relating to removal of nitrogen from sludge is known. However, recent work on advanced treatment of wastewaters provides some basis for considering possible nitrogen removal techniques.

Removal of nitrogen in the ammonia form is being used at the largest operative nitrogen removal installation - the South Tahoe Public Utility District in California (Slechta and Culp, 1967). Ammonia stripping was selected following pilot plant studies of microbial denitrification, ion exchange and stripping. Because ammonia exists in equilibrium with the ammonium ion, it is necessary to raise the pH to around 11 to get efficient removal in this gas transfer operation. An advantage of ammonia stripping in this particular case was that expense for pH adjustment was unnecessary since high lime concentrations were already in use for phosphorus removal. Estimated costs for 95 percent ammonia removal at Tahoe were 1.4¢/1000 gal. The Federal Water Pollution Control Administration (1968) estimates that the added cost for pH adjustment of waste treatment plant effluent to be about 3 ¢/1000 gal. without possible credit for lime recovery.

Removal of nitrogen by microbial denitrification involves biological oxidation to the nitrate form followed by creation of anaerobic conditions to cause denitrification and loss of gaseous nitrogen. In wastewater treatment schemes, reducing conditions required for denitrification are created by addition of raw waste to nitrified effluent (Johnson and Schroepfer, 1964) or by addition of an organic substrate such as methanol. The cost of nitrogen removal from waste treatment plant effluents by nitrification-denitrification has been estimated to be 5 to 6 ¢/1000 gal. (Federal Water Pollution Control Administration, 1968).

Nitrogen can be removed by converting it into organic nitrogen in the form of algae cells. The process is carried on in shallow ponds to allow harvesting of the algae. Success depends upon development of an economical means of separating algae and on creation of an outlet for the harvested product. Farrell, et al. (1968), estimated the cost of algae growth for 90 percent nitrogen removal from waste treatment plant effluent to be 4 ¢/1000 gal.

Other techniques for nitrogen removal include ion exchange, electrodialysis, reverse osmosis, chemical oxidation, and distillation (Eliassen and Tchobanoglous, 1968). Farrell, et al. (1968), estimated the cost of nitrogen removal from treated wastewater by ion exchange to be 8.1 ¢/1000 gal. Use of a natural zeolite, clinoptililite, for selective removal of ammonia by ion exchange is a possible variation of the normal ion exchange process (Mercer, et al., 1968). Electrodialysis of waste effluents has been estimated to cost 16 ¢/1000 gal. (Federal Water Pollution Control Administration, 1968). Distillation would cost several times this amount. Nitrogen removal by oxidation with chlorine may be accomplished for about 5.3 ¢/1000 gal. (Farrell, et al., 1968).

7.3.2 Removal Prior to Land Application

Several possibilities exist for reducing the nitrogen concentration before sludge is applied to land. They share the common limitation that the degree of nitrogen removal is restricted by the fact that part of the nitrogen exists as ammonia and part as organic nitrogen.

Ammonia nitrogen could be removed from digested sludge by stripping. Economical sizing of gas transfer equipment would require that the pH of the sludge be raised. Therefore, this method could be particularly advantageous if a high pH proved effective for disinfecting sludge. In areas where water softening is practiced, softening sludges could be beneficially used for pH adjustment.

Some slow loss of ammonia could be expected at pH values lower than those used in ammonia stripping operations. Loss of ammonia from storage lagoons used as a part of distribution facilities in land application systems would be expected, although data required to predict these losses are inadequate. Use of spray irrigation systems would be more advantageous than other distribution methods as far as nitrogen removal is concerned, because some volatilization of ammonia could be expected during sludge application.

Another interesting possibility would be to use chlorine for obtaining nitrogen removal as well as disinfection. Ammonia would be oxidized to nitrogen gas which would be lost to the atmosphere. The stoichiometric requirement for this oxidation is 6.25 mg chlorine/mg ammonia. However, this does not include allowance for reactions of chlorine with organic materials. Farrell, et al. (1968), noted that 9-10 mg of chlorine would be required per mg of ammonia if waste treatment plant effluent were chlorinated. Considerably more chlorine would be necessary for sludge treatment.

It would also be possible to biologically oxidize the ammonia in digested sludge to the nitrate form and then cause nitrogen gas evolution by biological denitrification. However, the cost would probably not be competitive with the other possible removal schemes, because of the great amount of aeration required to create conditions favorable for nitrification of anaerobic sludge.

7.3.3 Removal in Situ

Management practices for land disposal areas could be adjusted to minimize loss of nitrogen to the water environment. Some of these possible practices have the disadvantage of being susceptible to the whims of nature. For example, a rain at the wrong time could interfere with some of the management practices.

Nitrogen removal might be accomplished in situ by allowing nitrification to take place following sludge application and then to cause denitrification with the resulting evolution of nitrogen to the atmosphere. The reducing conditions required for denitrification might be brought about by flooding with sludge and/or by blocking drainage tile or ditches to raise the water table in the soil to cause anaerobiosis. Field scale data on the practicality of this scheme are not available. It does deserve consideration as an economical method for removing large amounts of nitrogen.

Additional management practices for land disposal systems which could reduce the severity of the nitrogen problem include the control of surface runoff and selection of optimum times for waste application.

7.3.4 Removal from Leachate

In areas where leachate from soils receiving wastes could be economically collected, nitrogen removal from leachate prior to release to the environment is a possibility. Advantages of treatment at this point would be that all of the nitrogen would be in the same form (nitrate) and interfering substances such as particulates and organic materials would have been removed.

Since the nitrogen would already exist in the oxidized form, denitrification would seem to be an economical means of nitrogen removal from leachate. A precedent for consideration of such a system has been set in studies of nitrogen removal from irrigation return flows in the San Joaquin Valley in California. Eliassen and Tchobanoglous (1968) have described McCarty's work on this project in which a carbon source (methanol) is added to the nitrified flow before it enters deep anaerobic ponds. After about 10 days retention to accomplish denitrification, the flow is aerated and discharged.

Absence of information on the chemical quality of leachate makes it difficult to speculate as to the applicability of algal synthesis systems for treatment of leachate. It is possible that lack of sufficient amounts of phosphorus or other nutrients would prevent the amount of synthesis required for effective nitrogen removal. Other nitrogen removal methods described in Section 7.3.1 could be used for treatment of leachate. However, it is not anticipated that they would be economically competitive with denitrification as a means of nitrogen removal.

7.3.5 Conclusions

There are a large number of possible methods to control the loss of nitrogen from waste applications to land. Unfortunately, sufficient information is not available at this time to recommend and estimate the cost of any particular procedure. However, present studies on nitrogen removal from wastewater indicate that this problem can be overcome through various management practices without incurring a heavy economic burden.

8.0 LAND DISTRIBUTION SYSTEMS

8.1 STATE OF THE ART

Numerous articles in the literature indicate that tank trucks, with gravity feed discharge ports in various boom arrangements, are commonly used for transportation and distribution of wastes on land (Scott, 1962 (Wolfel, 1967). Often, the truck capacities are only 1000 to 2000 gallons. Occasionally high pressure spray units, mounted directly on the tanker trucks or on trailers, are employed to spread the material. In addition, truck pumping of wastes thru light weight flexible pipe to sprinkler irrigation equipment has been tried. In general, transport of liquid waste by tanker truck and direct application on land is the most economical method of disposal when volumes are relatively small.

At San Diego, California, digested sludge has been successfully used for land reclamation since 1955. The material serves as a source of nutrients and as a soil conditioner for the vegetation of sterile dredged sands. Originally, the sludge was trucked to the disposal site and the sludge was applied in deep furrows which were later backfilled. Loadings as high as 600 tons per acre were applied (Dodson and Stone, 1962). In 1963, a 7 mile, 8" diameter pipeline was installed and sludge was then pumped directly into large shallow beds.

The New York City Park Department used high rate digested sludge to produce top soil for new parks (Scanlon, 1957). Marsh areas designated as potential park land were filled with garbage, covered with two feet of sand, and then treated with digested sludge. Tank trucks and barges were used to transport the digested sludge to a central manifold, where it was pumped through a distribution system to the designated areas. The sludge was applied by alternate flooding and drying until about 4 inches of topsoil material was produced from the sand and sludge mixture. After completion of the conditioning process, the areas were seeded to grass.

Several cities in the United States, as well as Europe, have operated farms for sludge utilization. Generall speaking, these operations have been conducted without any particular difficulties. Canton, Ohio has operated a municipal sludge disposal farm since 1900. However, because of the lack of proper farm management and the disposal of improperly digested sludges, local stream pollution occurred. This had led to demands that the practice be halted.

Law (1968) recently reviewed the literature and presented results of investigations dealing with land disposal of various wastes. Spray or sprinkler irrigation was reported as being usually used for the distribution of liquids containing a high content of organic waste.

8.2 APPLICATION RATES

Digested sludge may be utilized as a fertilizer for pasture and cultivated crops. When used as a fertilizer, one inch per year of a fresh 3 percent suspension will supply ample nitrogen and phosphorus for most crops. Generally, at this rate of application, carbon losses would equal carbon gains and organic residues would not accumulate.

In some cases, it may be desirable to minimize the number of acres used for sludge disposal. The rate at which carbon will be mineralized is expected to greatly vary with soil type, management practices and climatic condition. However, based on limited data, an organic accumulation is likely in areas like the humid midwest, when annual digested sludge applications exceed 10 to 12 inches per year on silt loam or finer textured soils.

With the continued application of digested sludge at rates that exceed 10 to 12 inches per year, a soil resembling a peat may be developed. This may be desirable as there are many productive peat soils. However, for solids accumulation to proceed very rapidly, the annual loading rate would have to be much higher than 10 to 12 inches per year. It appears that the organics that would accumulate in soils as a result of high loading rates would be stabilized at a fairly rapid rate by aerobic decomposition up to a total annual application of about 90 tons of solids per year. If the annual rate of application exceeds 90 tons per year (approximately 30 inches per acre per year), anaerobic conditions may prevail in fine textured soils and further stabilization may be slow.

Farm lands receiving heavy applications of digested sludge should be maintained at a pH of not less than 6.5. The establishment of proper drainage and control of the drainage water is essential to heavy sludge applications on all types of disposal areas. If subsurface drainage is not possible, digested sludge should be applied on level agricultural lands at rates not exceeding one to two inches per acre per year.

When digested sludge is applied on land at rates just sufficient to provide nitrogen and phosphorus fertility (about 1 inch per acre per year), a hundred foot wide strip of grass or other close growing crop should be established between the application area and waterways that receive the runoff. Where land is used for intensive disposal of digested sludge, detention ditches and reservoirs should be constructed to receive all runoff and discharge water. Runoff water collected in the reservoirs should be held long enough for solids to settle before decanted water is discharge. In some cases it may be necessary and/or profitable to use the stored water for irrigation of land areas outside of the disposal area. To minimize the size of reservoirs, all foreign water should be diverted from the disposal area or the area should be selected so that the only runoff and drainage water to be intercepted and held in the reservoirs is that which originated on the disposal area.

8.3 METHODS OF APPLICATION

8.3.1 Sludge

If digested sludge is frequently applied at heavy rates on sloping land, a high degree of control of drainage and runoff water from the area must be maintained to prevent stream pollution. A system of closed level-ridge terraces, filter bed drains, surface inlets, reservoirs, and irrigation equipment may be required to economically dispose of digested sludge and prevent water pollution. One such system is shown in Figure 8.3.1. Water treatment plant sludge could be similarly handled, either separately or included with the digested sludge.

On site lagoon storage will serve several purposes. First, it will allow for continual operation of the conveyance system. Also, with construction of multiple lagoons, digested sludge irrigation could be alternated in such a fashion as to allow for a minimum period of ten days storage time to increase pathogen die-away and losses of ammonia nitrogen by volatilization. Thus, two potential water pollution hazards, excessive concentration of pathogenic organisms and nitrate nitrogen may be controlled.

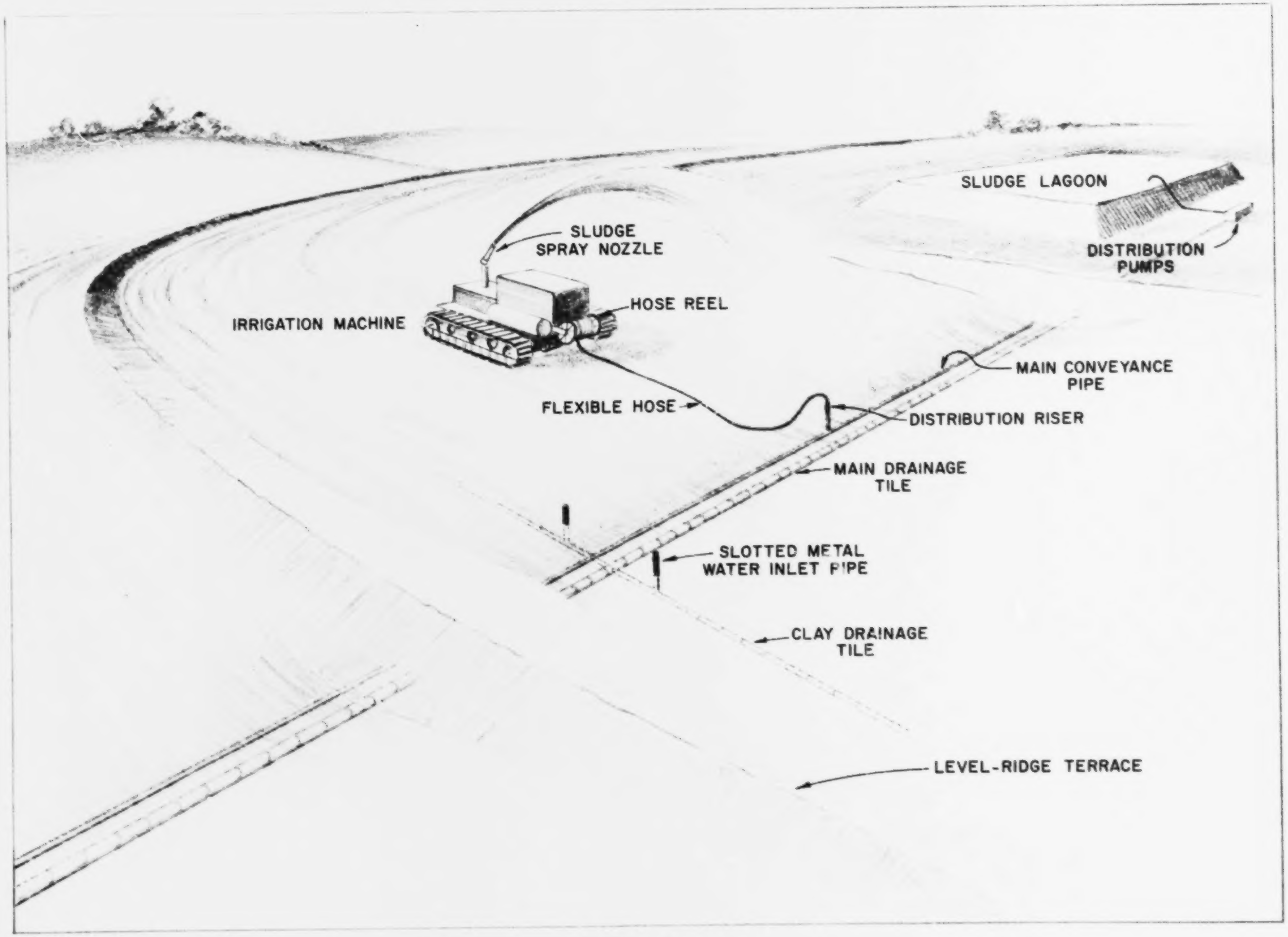

SLUDGE LAGOON
DISTRIBUTION PUMPS
SLUDGE SPRAY NOZZLE
HOSE REEL
IRRIGATION MACHINE
MAIN CONVEYANCE PIPE
FLEXIBLE HOSE
DISTRIBUTION RISER
MAIN DRAINAGE TILE
SLOTTED METAL WATER INLET PIPE
CLAY DRAINAGE TILE
LEVEL-RIDGE TERRACE

FIGURE 8.3.1
PROPOSED DISTRIBUTION SYSTEM
FOR SLUDGE WASTE MANAGEMENT

A series of level-ridge, tile-outlet terraces will prevent stream pollution by high organic loads. The terraces should be constructed according to U. S. D. A. Soil Conservation Service specifications, except that storage capacities should be increased, especially if loading rates of waste are initially maximized for the purpose of land reclamation. Slotted metal water inlet pipes will be needed as part of the tile-outlet system to prevent excessive over-topping of the terraces during periods of rainfall which are exceedingly intense or of long duration. These inlet pipes should be supplemented by the installation of about 50 feet of 4 or 5 inch clay drainage tile joined perpendicular to and on each side of the main drainage tile outlet. The 100 feet of farm drainage tile installed in the terrace channel should be backfilled to the surface with bank-run gravel. The slotted metal surface inlet pipes will be at such an elevation that they will be operative only during an abnormal high water retention period. The water from waste application and normal precipitation will be removed by infiltration and percolation through gravel to the clay farm drainage tile.

The storage capacity of level-ridge terraces should be increased over the current design for holding 2 inches of runoff from the watershed. The freeboard of the terraces will need to be adjusted to handle all runoff water after the bed elevation has been increased by 6 to 12 inches of applied waste residues and to hold the additional water supplied during waste applications.

Detention reservoirs will be constructed to impound the effluent from the tile and surface-inlet drainage system. If the quality of the impounded drainage water is too poor to be released to streams or constructed waterways, it will be pumped to the sludge storage lagoons for reuse, or used to irrigate areas other than the disposal area.

The distribution or irrigation system will consist of pumps to move the waste materials from the storage lagoons, the main conveyance pipe to the disposal area and a self-propelled irrigation machine which will be supplied by means of a flexible hose. The main conveyance pipe will be installed with and adjacent to the main drainage tile outlet line. A riser will be installed in the conveyance pipe about midway between terraces with a key and valve arrangement on the riser for connecting the flexible irrigation hose. The sprinkler irrigation machine will be equipped with a pump capable of producing sufficient head to provide the desired spray distribution from a single nozzle-gun. The pump for the main conveyance line will maintain sufficient suction pressure for the second pump on the irrigation machine so as to prevent collapse of the flexible hose.

A powered hose reel would be mounted on the irrigation machine for use in transporting the flexible hose. The hose could possibly be 700 feet long. With the riser located midway between terraces, this would allow an uninterrupted travel of nearly 1400 feet for the irrigation machine during a single run. The nozzle-gun will be adjusted so that it will traverse back and forth over an arc length just sufficient to provide a uniform application of sludge over the soil surface between the upper terrace back-slope and the lawn terrace front-slope. The ground speed of the self-propelled irrigation machine will determine the rate of waste application for a given terrace spacing. These spacings will vary with slope and 12 percent is considered to be a practical maximum.

For a 1 inch diameter nozzle-gun, the irrigation machine will require a centrifugal pump capable of delivering about 500 GPM at a pressure of about 100 PSI. Working ground speeds should be selectable over a range of 0 to 20 feet per minute, with a transporting speed of about 5 MPH.

Although such an irrigation machine does not presently exist, a resemblance is commercially available. In Hawaii, large irrigation machines have recently been used in the sugar and pineapple fields. These machines are reported to be capable of spraying up to 2000 GPM through 2000' of 8" hose, and have been utilized on terrain varying from ocean beach to hill sides. Still larger machines are being planned.

If digested sludge was used to upgrade well drained marginal farm land or to reclaim strip mined lands, it is probable that maximum amounts of material would be applied during the first year or two with little regard to the establishment of vegetation. The major goal would be to supply nutrients and improve soil structure and water retention conditions that favor the production of crops or vegetative cover, while protecting the quality of water resources in the area. Later, as more attention was given to revegetation or crop production, rates and frequencies of waste applications would be reduced to those commensurate with optimum plant growth or production. It is possible that as much as about 30 inches per acre (approximately 90 tons of sludge solids) might be applied during the first one or two years under normal midwest climatic conditions. After vegetation was established on the disposal area, annual applications of less than 10 inches per acre may be called for. Rates of application after the initial high rates for land upgrading or reclamation would depend on land usage and priorities placed on use of waste materials for improving other areas.

8.3.2 Dredgings

Maintenance dredgings, after sand removal, could be used to upgrade or improve marginal sandy, shallow or stony soils. Dredgings might also be used to reclaim stony or acid mine spoil banks. A few inches of silt loam or silty clay loam dredging materials would greatly increase the moisture holding capacity of coarse textured and/or stony strip mine spoil banks.

The application of dredgings on land would require about the same kind of terrace and other structures for the control of runoff and drainage water as outlined above for sludge disposal. With the sand removed, the same type of distribution system as described for sludge would be suitable for uniformly spreading dredgings over the land surface.

If dredgings are applied as a suspension containing 20 percent solids by weight, then more than ten inches of liquid will be needed to uniformly deposit an inch of solids over the soil surface. A one foot layer of dredging solids over the surface of acid strip mine spoil banks should be sufficient to prevent further acid production by oxidation of pyritic materials associated with coal and coal-bearing strata. If weather conditions limit application rates to a maximum of 30 inches per year, at least four or five years will be required to build-up this one foot depth of solids.

Following the application of dredging materials to improve soil moisture holding capacities of coarse materials, digested sludge could be applied to provide water and nutrients for established vegetation. On droughty areas, the rate and frequency of sludge applications would be governed by the need for additional moisture. Water might be supplied to droughty areas from the detention reservoirs constructed in maximum sludge disposal sites.

8.3.3 Fly Ash

High fly ash loading rates could add considerable dissolved solids to the groundwater. However, fly ash applied at 400 to 600 tons per acre would be very valuable in the reclamation of acid strip mine spoil banks. Like dredgings, fly ash may be used to increase the moisture holding capacities of coarse textured and stony soils or strip mine spoil banks.

Although fly ash used at sufficiently high rates may provide for the neutralization of acid conditions, it would supply very little in the form of available plant nutrients. Thus, its greatest value as a resource would be obtained by a combination with dredgings and/or digested sludge.

8.4 CONCLUSIONS

Because a high degree of land leveling would not be required for applications of waste by either truck or sprinkler irrigation systems, it is likely that these surface application methods would receive prime consideration over ridge and furrow systems. In the case of strip mine spoil banks, further land disturbance would be particularly undesirable because of increased exposure of pyritic materials to oxidizing conditions. If as much as 12 inches per acre per year of liquid waste is applied, a sprinkler system would probably be chosen. However, the actual annual application rate is likely to be considerably less, except possibly for the first few years. Therefore, an irrigation machine system could prove advantageous. Machines, similar to those required for waste disposal, are commercially available.

The management of waste materials by land disposal is entirely feasible. However, it is realized that numerous problems could occur in establishing a regional system for disposal of various waste materials. Therefore, each step in an initial investigation of disposal methods should be approved by a planning board which represents all interests in the affected region. The investigation should be primarily concerned with a complete hydrologic study of possible effects of waste disposal on the quantity and quality of water resources in the area.

9.0 OCEAN DISTRIBUTION SYSTEMS

9.1 STATE OF THE ART

Disposal of digested sludge at oceanic sites is being carried out in the United States using two different basic systems. Barge and tanker transport and release along a course through the disposal area is practiced along the east coast (New York and Philadelphia). Pipe transport and release through a diffuser outfall is being used at Los Angeles, San Diego and Seattle. Selection of pipe and diffuser disposal for use by communities on the west coast has been favored by the existence of the following conditions:

1. Deep water (200-300 feet) disposal sites at distances of a few miles from shore.
2. Disposal sites close to the source of sludge.
3. Operation of a few large treatment plants, rather than many scattered small treatment plants.

The use of barge treatment and release from the barge for disposal on the east coast has been favored by nearly the converse of the factors on the west coast, that is:

1. Water depths of 100 feet or less along the coast out to distances of 25 miles or more.
2. Communities are located some distance from the ocean.
3. Several treatment plants within a community may be served by a single barge.

Disposal by barge was considered in the report by the Municipality of Metropolitan Seattle (1965) to be disadvantageous in that "discharge from a barge essentially at the surface, would result in very poor dispersion and dilution". However, this statement was made with reference to diffuser discharge of sludge "prediluted" by 1 to 100 with treatment plant effluent and without a detailed analysis of the barge disposal case.

Some of the features of barge disposal are:

Advantages

1. Flexibility in operation. Disposal area may be changed without great economic penalties.
2. Initial dilution can be altered by changing the length of the run during discharge.

3. Relatively low capital cost.

Disadvantages

1. Material is added to surface layer and is visible for various time periods.
2. Location of disposal point is dependent on the accuracy of navigation and conscientious conduct of the disposal operations.
3. Disposal operations may be weather-limited.
4. Relatively high operating costs.

The pipe and diffuser technique of disposal has these features:

Advantages

1. Material is added at the bottom of the water column. Under conditions of favorable density structure in the ocean, proper diffuser design avoids contamination of surface layers.
2. Disposal point is known and quality control of disposal operation is simple.
3. Relatively low operating cost.

Disadvantages

1. Lack of flexibility. Change in disposal site requires review of pumping capacity and laying of new pipes or extensions.
2. The initial dilution that can be achieved is determined by the site characteristics, with flexibility to the extent of practical diffuser lengths (a few thousand feet).
3. Relatively high capital cost.

9.2 ATLANTIC COAST ALTERNATIVES

The environmental aspects of the two systems (barge-tanker and pipe-diffuser) on the east coast may be compared by an example of hypothetical systems serving a given size population. Consider a population unit of 1.5 million people with a digested sludge production rate of approximately 0.15 pounds per day per capita or 1.0×10^5 cubic feet per day of a 3.5% digested sludge suspension. In Section 6 of this report, various environmental aspects of sludge disposal were reviewed and a desirable dilution factor of 1 to 3×10^4 was suggested - based on a 3.5% digested sludge suspension. With higher or lower concentrations, the desirable dilution would change in direct proportion to the initial concentration.

For this analysis, the following disposal site characteristics are assumed:

1. Water depth of sixty feet.
2. Nontidal drift of 5 cm per sec (0.1 knots).
3. A rotary tide with '5 cm per sec north-south component and 10 cm per sec east-west component (typical values as presented by Haight, 1942).
4. A thermocline centered at a depth of thirty feet.

9.2.1 Barge and Tanker Disposal

New York City is presently using tankers with capacities ranging from 0.4×10^5 to 1×10^5 cubic feet. A tanker size of 1×10^5 cubic feet will be taken as an example. If this tanker was unloaded along a course length of 33,000 feet (six nautical miles) and the sludge suspension was spread across the wake to a width of 33 feet and mixed vertically to a depth of 20 feet (assuming tanker draft of 15 feet), the initial dilution would correspond to mixing with 2.2×10^7 cubic feet of ocean water. This would provide an initial dilution of 1 to 220 and the nominal initial concentration of suspended solids would be 160 mg per liter.

The resulting patch of contaminated water would undergo further dilution as it drifted with the local current velocity. Dilution during the drift might be described by the following equation developed by Okubo (1968):

$$\frac{CD}{M} = \frac{1}{4ab}\left[\operatorname{erf}\left(\frac{x-ut+\frac{1}{2}a}{wt}\right) - \operatorname{erf}\left(\frac{x-ut-\frac{1}{2}a}{wt}\right)\right]\left[\operatorname{erf}\left(\frac{y-vt+\frac{1}{2}b}{wt}\right) - \operatorname{erf}\left(\frac{y-vt-\frac{1}{2}b}{wt}\right)\right]$$

where:

C = concentration
D = depth
M = quantity discharged
a = the smaller horizontal dimension of initial patch
b = the larger horizontal dimension of initial patch
x = horizontal space coordinate
u = velocity in the x direction
y = horizontal space coordinate
v = velocity in the y direction
w = diffusion velocity

The observed values of w in various oceanic environments range from 0.1 to 1.3 cm per second, and a value of 0.5 cm per sec is taken here. Under the above assumptions and with an initial concentration of 160 mg per liter, the maximum (peak) concentration in the center of the patch would vary as follows with time:

1 hour	48 mg per liter
2 hours	26 mg per liter
4 hours	12.8 mg per liter
8 hours	6.4 mg per liter
16 hours	3.2 mg per liter
24 hours	2.1 mg per liter
48 hours	1.0 mg per liter
72 hours	0.7 mg per liter

In Section 6, it was suggested that a permissible increase in total suspended solids might be taken as 1 ppm without a clearly visible change in transparency. Thus it can be seen that the duration of the "hot spot" from this barge disposal example is of the order of two days. Favorable oceanographic conditions might produce a w of 1 cm per second and the duration over a 12-mile course rather than the assumed 6-mile corse and the "hot spot" duration would be one day under average oceanographic conditions. In any case, if a one to two day duration of the "hot spot" is taken as a minimum design criterion, initial loading of the patch must be approximately as assumed above.

Two considerations support the selection of this duration criterion. First, the patch may drift into recreational areas. Haight (1942) reports that observed surface currents in knots were 0.014 times the wind speed in statute miles per hour. With a wind speed of 10 miles per hour, an average figure for many localities, the surface current would be 0.14 knots or 3.4 nautical miles per day. With a disposal site ten miles offshore, the dilution criterion of 3×10^4 would be met before the material drifts into beach areas. Higher initial loading would require disposal further offshore to avoid beach water degradation. Second, longer persistence of the "hot spot" might be expected to stimulate algal growth. There does not seem to be any information on this subject in the literature. A rough notion of permissible duration of the "hot spot" is inferred from the growth lag times in cultures of several days and the division rates of approximately one day (Lanskaya, 1961) observed for marine algae. However, it should be noted that some species have maximum division rates under optimum conditions that result in doubling in 12 hours or less.

The practical consequences of this 48 hour, hot spot duration criterion do not seem great for the example above, i. e., recommended use of a vessel of 1×10^5 cubic feet capacity and a requirement of a daily trip. However, for schemes proposing higher sludge concentrations and larger vessels, considerably longer discharge lengths will be required. As an example, for disposal of 7.5% sludge suspensions by barges with a capacity of 2.5×10^5 cubic feet, discharge along a 32 mile course is indicated to be necessary.

9.2.2 Pipe and Diffuser Disposal

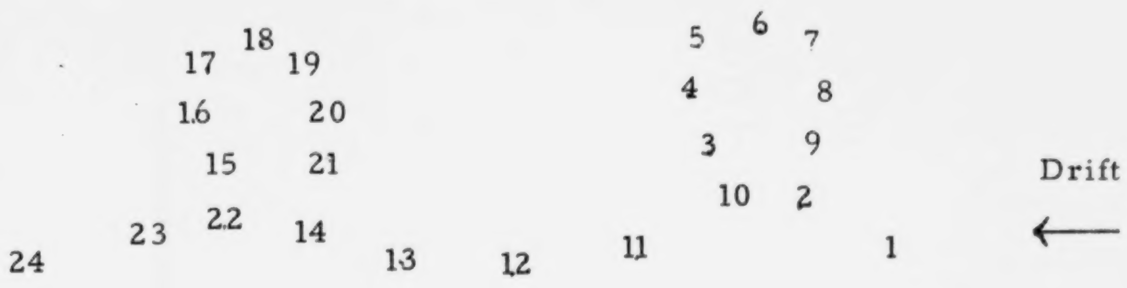
The same population unit of 1.5 million people also might be served by a pipeline and diffuser outfall. Many east coast areas have water depths of [illegible] feet within seven miles of the shoreline and such a site is considered here for comparison with barge disposal. The first stage dilution that occurs as the buoyant sludge suspension rises from outlets on the bottom depends primarily on the nozzle diameter and Froude number. Much of the recent experimental and theoretical work on the dilution during this step has been summarized by Abraham (1963). Two and one-half inch diameter nozzles are being used successfully with the Los Angeles County outfall at Whites Point. For this example, two-inch diameter sludge nozzles are assumed, although smaller diameter nozzles would improve the first stage dilution.

For a diffuser of 8 nozzles spaced at 140 feet (total length 1000 feet ±), the flow per nozzle would be 0.15 cfs to discharge the 1×10^5 cubic feet per day of 3.5% sludge suspension in this example and the Froude number would be 36. In the absence of significant density stratification (no thermocline), the discharge would reach the surface and, in the absence of recirculation in the surface layer, the initial dilution would be approximately 1 to 300, and the nominal concentration of suspended solids would be 117 mg per liter. As the eight individual patches move with the surface current, they will blend together. Experience with diffuser outfalls of this type has shown that the field from a diffuser of the dimensions described will be essentially uniform at a distance of 2500 feet down current from the diffuser (J. H. Carpenter, personal communication). Tidal currents on the shelf have speeds of 0.2 to 1 knots. Assuming a speed of 0.3 knots, the flow through a section with a width of 1000 feet and a depth of 10 feet would be 5.1×10^3 cfs. The resulting dilution would be roughly 4×10^3, and this would produce a uniform field with a suspended solids concentration of 9 mg per liter.

This high concentration area (1000 by 2500 feet) would be present continuously and is difficult to compare with the patches produced by the barge operation. The latter would have a lifetime of 6 hours at this concentration or higher but with dimensions of six miles by roughly 300 feet. A further aspect of the diffuser case is that in many localities, the discharge will not reach the surface from May through October. The specific gravity of a 3.5% sludge suspension would be approximately 1.014 if the sludge were 60% mineral and 40% organic. The specific gravity of surface waters on the shelf 5 to 10 miles offshore is roughly 1.022 and that of waters at sixty feet is roughly 1.025 (Woods Hole Oceanographic Institution, 1961). Therefore, a three to one mixture of deeper water with the sludge suspension is more dense than the surface water. A typical density profile of the shelf waters shows nearly uniform density in the upper 20-30 feet, a rapid increase in density in the next layer, and nearly uniform density in the deeper water.

Thus, even in the relatively shallow areas of the shelf, it is possible to avoid contamination of the surface layers during the summer and early autumn. Assuming a vertical mixing throughout a layer with a thickness of ten feet, the concentrations in the field located at mid-depth would be approximately those calculated above, i. e., 9 mg per liter.

If the field moves with the mean current to produce a long, straight plume, and assumed values of 0.5 cm per sec for diffusion velocity and 0.1 knots for drift rate are taken, the solution to the steady-state diffusion equation for a continuous vertical place source of contaminant (Okubo, 1968) shows a ten fold reduction in concentration would occur at a distance of ten miles downcurrent from the 1000 foot diffuser. This result underestimates the dilution with distance because during the 100 hours required for the field to drift the ten miles, it is subjected to the rotary tidal currents that are stronger than the average drift current. The trajectory of the plume will not be straight and more rapid dilution than that calculated from the simple model would be expected. The pattern of the center of mass would be as follows:



Age in Hours

There does not seem to be any experimental evidence on the rate of dilution with distance under these conditions. The conservative calculation will be used here but it should be emphasized that it is conservative to an unknown extent.

In summary for the example discussed above, the barge technique of disposal produces daily patches six miles in length that require two days to dilute to the environmental criterion, i. e., two patches are present near the disposal site at undesirable concentrations and the affected area is roughly 12 miles by a few thousand feet. For the pipe and diffuser technique, an area of 10 miles by a few thousand feet is computed to be above the desirable dilution.

9.3 CONCLUSIONS

In view of the comparable areas that are degraded by the two techniques, environmental considerations do not lead to a clear cut choice. Avoidance of surface layer contamination during the summer may favor pipe and diffuser disposal at some sites. However, as increased loading develops in some areas, it will be necessary to move the disposal sites further offshore, which could favor barge disposal. A principal point of uncertainty is the large scale "turbulent diffusive" processes that lead to a renewal of nearshore waters with water from further offshore, which determines the general area assimilative capacity. If the only renewal process was the nontidal drift of 0.1 knots, each population unit of one and one-half million people would require a 2 mile section perpendicular to the coast to obtain a desired 3×10^4 dilution. The sludge from a population of 15 million people would require the diluting capacity of the first twenty miles. Therefore, the random processes that lead to renewal of the nearshore waters must be evaluated before a rational program of sludge disposal can be developed.

APPENDIX A: WASTE CHARACTERISTICS

DIGESTED SLUDGE

Properties of digested sludge reflect the nature of the waste being treated, the type of treatment, and the efficacy of the digestion system. The general characteristics given here are for a typical anaerobically digested mixed primary and waste activated sludge from a municipal waste-water treatment plant. Values refer to the total digester output, i.e. without supernatant withdrawal.

Physical Properties

Solids Concentration	3.0 - 3.5% by wt. ±
Other Properties	Musty odor, black color, homogenous with smooth texture.

Chemical Properties

pH	near 7.0
Alkalinity	2000 - 6000 mg/l as $CaCO_3$
Volatile Acids	Less than 500 mg/l as acetic acid
Total Nitrogen	1500 mg/l ±
Phosphorus	800 mg/l ±
Potassium	150 mg/l ±
Heavy Metals	Concentrations are highly variable - usually minimal from pure domestic waste-water treatment operations, but high concentrations of various heavy metals can result if wastes of industrial origin are included.

Biological Properties

Pathogenic Micro-organisms & Helminths	Appreciable reduction achieved by digestion process, but low concentrations will be present.
Viruses	Little information available, but all known reported attempts to isolate viruses have been negative.

MAINTENANCE DREDGINGS

Generally, few analyses have been made of maintenance dredgings in the Atlantic Coast Region. However, a complete characterization of sediments from eight harbors in the Great Lakes area was presented in a recent report submitted to the U. S. Army Corps of Engineers by the Center for Great Lakes Studies (University of Wisconsin, 1968). Results of this report are the primary basis for the values given below:

Physical Properties	
General Description	Mostly silt and clay
Dry Density	0.77 to 1.45 gm/ml
Particle Size	95% less than 2 mm
Other Properties	Light gray to reddish brown or black color, sometimes odorous.
Chemical Properties	
COD	20 - 350 mg/g
BOD	up to 13 mg/g
Volatile Solids	14 to 184 mg/g
Oils & Grease	up to 40 mg/g
Total Nitrogen	0.1 - 6.9 mg/g
Phosphorus	0.4 - 4.0 mg/g
Iron	up to 96 mg/g
pH	Usually around 7.0 (6.3 to 11.0 overall range)
Biological Properties	Highly variable depending upon degree of harbor pollution.

FLY ASH

Properties of fly ash are dependent upon the coal combustion operating conditions. Both the age and efficiency of a boiler affect the size distribution of the residual fly ash produced. Coarser fractions tend to be composed of carbonaceous particles which result from incomplete combustion of the coal, and agglomerated ash particles which result from sintering.

Physical Properties

Particle Size	65% passing 325 mesh 95% passing 100 mesh
Mean Density	1.9 to 2.5 g/cm^3
Other Properties	Usually spherical particles, gray color, normally solid, crystalline in nature.

Chemical Composition

SiO_2	40 - 70 %
Al_2O_3	20 - 35
Fe_2O_3	0.5 - 25
C	0.5 - 15
CaO	0.4 - 8
MgO	0.3 - 5
Na_2O	0.1 - 2
K_2O	0.4 - 5
TiO_2	0.7 - 2
Mn_3O_4	0 - 0.2
SO_3	0 - 2.5
P_2O_5	0 - 0.8 %

In addition, a large number of micro-elements may be contained in fly ash. Only manganese and titanium are liable to be present in concentrations greater than 0.1%. Boron, chromium, copper, lead, nickel and zinc would probably fall within the range of 0.01 to 0.1 percent.

WATER TREATMENT PLANT RESIDUES

Since aluminum and iron salts are generally used to accomplish coagulation, hydrated aluminum and iron oxides are the chief constituents of sludges from water treatment plants practicing chemical coagulation. In addition to these hydrates, the residues contain entrained particulate matter, mostly inorganic and consisting of fine sands, silts and clays. In the absence of gross pollution of the raw water source, the entrained or absorbed organic fraction of the residue is small and for the most part, stable or non putrescible.

The solids content of the filter washwater is generally low and reflects the efficiency of the pretreatment while the composition of the solids is generally similar to that of the other solids produced. Hydrated oxides of aluminum, iron, manganese, and magnesium; carbonates of calcium and iron; plus silicate materials form the largest fraction of the solids. Organic matter may be present in the form of algae, plankton, slimeforming bacteria, and spent activated carbon. Since rapid sand filters can often support a luxurious biological growth. the filter backwash may contain a far larger fraction of organic matter than is found in the solids from sedimentation units. Moreover, the bacterial count of filter backwash waters can be quite high.

TYPICAL PROPERTIES

	Aluminum Coagulation	Filter Backwash
% Solids	0.2 - 2.0	.01 - 0.1
Color	gray-brown	varies widely: gray-brown-black
Odor	none	none
BOD, mg/l	30 - 300	10
COD, mg/l	300 - 5000	100
Settleability	50% in 8 hrs.	80% in 2 to 24 hrs.
Filterability, specific resistance	poor, 0.2×10^{-10}	poor
Viscosity, g/cm-sec	0.03	0.01 (water)

3.0

REFERENCES: WASTE QUANTITIES

3.1 Digested Sludge

ASCE, 1959, "Sewage Treatment Plant Design", *ASCE Manual of Engineering Practice*, No. 36, p. 6.

Fair, G. M. and Geyer, J. C., 1954, *Water Supply and Waste-Water Disposal*, p. 582

Federal Water Pollution Control Administration, 1968a, *The Cost of Clean Water*, Volume II, pg. 3-6, 72.

Federal Water Pollution Control Administration, 1968b, *Summary Report Advanced Waste Treatment*, July 1964 - July 1967.

Harza Engineering Co., 1967, *Land Reclamation Project*, pg. I-1.

Loehr, Raymond C., 1968, "Variation of Wastewater Quality Parameters", *Public Works*, May.

U. S. Bureau of the Census, Current Population Reports, Series P-25
No. 371 "Estimates of the Population of SMSA's, July 1, 1965".
No. 378 "Provisional Estimates of the Population of the Largest Metropolitan Areas: July 1, 1966".
No. 375 "Revised Projections of the Population of States: 1970 to 1985".

Watson, K. S. and Clarke, S., 1961, "What Ground Garbage Does to Sewage Systems", *American City*, 76, 7.

3.2 Maintenance Dredgings

U. S. Army Corps of Engineers, 1962 through 1966, *Annual Report of the Chief of Engineers*.

U. S. Army Corps of Engineers, 1968, "Harbor and Port Development", p. 7.

U. S. Army Corps of Engineers, 1969, "Dredging and Water Quality Problems in the Great Lakes".

3.3 Fly Ash

Brackett, C. E., 1967, "Availability, Quality and Present Utilization of Fly Ash", *Bureau of Mines Info. Circ. 8348*, pg. 16.

Jarrige, A., 1967, "An Attempt to Explain French Success in the Utilization of Fly Ash", *Bureau of Mines Info. Circ. 8348*, pg. 142.

3.4 Filter Plant Residues

Durfor, C. N. and Becker, E., 1962, "Public Water Supplies of the 100 Largest Cities in the United States, 1962", Geological Survey Water Supply Paper 1812.

Hudson, H. E. Jr., 1968, "How Serious is the Problem", Proceedings Tenth Sanitary Engineering Conference, University of Illinois.

Murray, C. R., 1968, "Estimated Use of Water in the United States, 1965", Geological Survey Circular 556.

4. 0

REFERENCES: PRESENT METHODS & COSTS OF DISPOSAL

4. 1 Digested Sludge

Burd, R. S., 1968, "A Study of Sludge Handling and Disposal", Federal Water Pollution Control Administration, Water Pollution Control Series Publication No. WP-20-4, 369 pp.

Connell, C. H. and Garrett, M. T. Jr., 1963, "Disinfection Effectiveness of Heat Drying of Sludge", J. Water Pollution Control Federation, Oct. 1963, 1262-1268.

Frankel, J. I., 1966, "Incineration of Process Wastes", Chem Eng., 73, 18, 91.

Riddell, M. D. R. and Cormack, J. W., 1966, "Ultimate Disposal of Sludge in Inland Areas", Paper presented at 39th Annual Meeting of Central States Water Pollution Control Association, Eau Claire, Wisconsin.

Robertson, J. H. and Woodruff, P. H., 1967, "Incineration - The State of the Art", Water and Sewage Works, 114, RN, R146.

4. 2 Maintenance Dredging

Cable, C. C., 1969, "Optimum Dredging and Disposal Practices in Estuaries", Journal of the Hydraulics Division, ASCE, Vol. 95, HYI, Proc. Paper 6343, pg. 103-114.

Mauriello, L. J. and Caccese, L., 1963, "Hopper Dredge Disposal Techniques and Related Development in Design and Operation", U. S. D. A. Agr. Res. Ser. Misc. Pub., 970, 598-613.

U. S. Army Corps of Engineers, 1967, "Analysis of Consolidated Statement of Operations Seagoing Hopper Dredges, Fiscal Year 1966".

U. S. Army Corps of Engineers, 1969, "Dredging and Water Quality Problems in the Great Lakes", Buffalo District.

4.3 Fly Ash

None

4.4 Filter Plant Residues

Dean, J. B., 1953, "Disposal of Wastes from Filter Plants and Coagulation Basins", *Jour. AWWA*, *45:11*, 1229-1237.

Remus, G. J., 1966, "Joint Discussion: Waste Disposal - Water Treatment Plants", *Jour. AWWA*, *58*, 1112.

5.0

REFERENCES: ENVIRONMENTAL CONSIDERATIONS FOR REGIONAL LAND DISPOSAL SYTEMS

5.1 Effect on Water Environment

Allison, F. E., 1955, "The Enigma of Soil Nitrogen Balance Sheets", Advances in Agronomy (A. G. Norman, editor), Academic Press, Inc., New York, 213.

Allison, F. E., 1966, "The Fate of Nitrogen Applied to Soils", Advances in Agronomy (A. G. Norman, editor), Academic Press, New York, 210-258.

Anderson, M. S., 1956, "Comparative Analysis of Sewage Sludges", Sewage and Industrial Wastes 28, 2, 132-141.

Bailey, G. W., 1968, "Roles of Soils and Sediment in Pollution Control - Part I, Reactions of Nitrogenous and Phosphatic Compounds with Soils and Geologic Strata", Federal Water Pollution Control Administration, 90 pp.

Bears, J. K., 1961, "Experiences in the Netherlands with Contamination of Ground Waters", Proceedings of Symposium on Ground Water Contamination, U.S. Public Health Service, R. A. Taft Sanitary Engineering Research Laboratory Report No. 67-11, 201 pp.

Boone, L. V., 1969, "N, P, K, and pH for Corn", Proc. 1969 Illinois Fertilizer Conference, University of Illinois, Urbana, 16-17.

Burden, E. H. W. J., 1961, "The Toxicology of Nitrates and Nitrites with Particular Reference to the Potability of Water Supplies", The Analyst, 86, 420.

Caldwell, E. L., 1938, "Pollution Flow from a Pit Latrine When Permeable Soils of Considerable Depth Exist Below the Pit", J. of Inf. Diseases, 62, 3, 225.

Carlson, G. F., et al., 1968, "Virus Inactivation on Clay Particles in Natural Waters", J. Water Poll. Control Fed., 40, R89.

Collison, R. C. and Mensching, J. E., 1930, "Lysimeter Investigation: I Nitrogen and Water Relation of Crops in Legume and Non-Legume Rotation", Technical Bul. #116, Cornell Univ.

Drewry, W. A. and Eliasen, R., 1967, "Virus Movement in Ground Water", Paper presented at the 40th Annual Conference, WPCF, New York.

Engelbrecht, R. S. and Morgan, J. J., 1961, "Land Drainage as a Source of Phosphorus in Illinois Surface Waters", Algae and Metropolitan Wastes, U.S. Public Health Service, San. Engr. Center Publication TR W 61-3, 74-79.

Geldreich, E. E., Huff, C. B., Bordner, R. H., Kabler, P. W., and Clark, H. F., 1962, "The Faecal Coli-aerogenes Flora of Soils from Various Geographical Areas", J. Appl. Bacteriol., 25, 87-93.

Geldreich, E. E., 1966, "Sanitary Significance of Fecal Coliforms in the Environment", Federal Water Pollution Control Administration, Water Pollution Control Research Series Publication WP 20-3, 121 pp.

Gilcreas, F. W. and Kelly, S. M., 1955, "Relation of Coliform-Organism Test to Enteric-Virus Pollution", J. American Water Works Assoc., 47, 683.

Grim, R. E., 1953, Clay Mineralogy, McGraw-Hill, New York.

Hallgren, R., 1942, "Epidemic Hepatitis in the County of Vasterbotten in Northern Sweden - An Epidemiological and Clinical Study", Acta. Med. Scand. Suppl., 140.

Hanks, T. G., 1967, "Solid Waste/Disease Relationships", Pub. Health Service Pub. No. 999-UIH-6, U. S. Dept. Health, Education and Welfare, Washington, D. C., 179 pp.

Harmeson, R. H. and Larson, T. E., 1969, "Interim Report on the Presence of Nitrates in Illinois Surface Waters", Proc. 1969 Illinois Fertilizer Conference, University of Illinois, Urbana, 33-36.

Henderson, J. M., 1962, "Agricultural Land Drainage and Stream Pollution", J. San. Engr. Div., ASCE, 88, SA6, 61-74.

Jenkins, S. H. and Cooper, J. S., 1964, "The Solubility of Heavy Metal Hydroxides in Water, Sewage and Sewage Sludge - III The Solubility of Heavy Metals Present in Digested Sewage Sludge", Int. J. Air Wat. Poll., 8, 695-703.

Jenne, E. A., 1968, "Controls on Mn, Fe, Co, Ni, Cu, and Zn Concentrations in Soils and Water: the Significant Role of Hydrous Mn and Fe Oxides", Trace Inorganics in Water, Advances in Chemistry Series 73, American Chemical Society, Washington, D. C., 337-387.

Krone, R. B., Orlob, G. T., Hodgkinson, C., 1958, "Movement of Coliform Bacteria Through Porous Media", Sewage and Industrial Wastes, 30, 1, 1-14.

Mackenthum, K. M., Ingram, W. m. and Porges R., 1964, "Limnological Aspects of Recreational Lakes", U. S. Public Health Service Publication No. 1167, 176 pp.

Mallmann, W. L., and Mack, W. N., 1961, "Biological Contamination of Ground Water", Proceedings of Symposium on Ground Water Contamination, U. S. Public Health Service, R. A. Taft Sanitary Engineering Center Technical Report W 61-5, 218 pp.

Mann, H. H. and Barnes, T. W., 1957, "The Permanence of Organic Matter Added to Soils", Jour. of Agr. Sci., 48, 160-163.

McCarty, P. L., et al., 1967, "Sources of Nitrogen and Phosphorus in Water Supplies - Task Group Report", J. Am. Water Works Association 59, 3, 344-366.

McGauhey, P. H. and Krone, R. B., 1967, "Soil Mantle as a Wastewater Treatment System", University of California Sanitary Engineering Research Laboratory Report No. 67-11, 201 pp.

Millar, C. E., 1955, "Soil Fertility", John Wiley and Sons, New York, 436 pp.

Morgan, M. F. and Jacobsen, H. G. M., 1942, "Soil and Crop Interrelations of Various Nitrogenour Fertilizers, Conn. Agr. Exp. Sta. Bull., 458.

O'Connor, J. R. and Morris, J. A., 1955, "Recovery of Texas-1 Type Coxsackie Virus from Blood of Wild Rabbit and from Sewage Contaminating Rabbits' Feeding Ground", Am. J. Hyg., 61, 314.

Ohle, W., 1953, "Phosphorus as the Initial Factor in the Development of Eutrophic Waters", Vom Wasser, 20, 11-23.

Parizel, R. R., et al., 1967, "Waste Water Renovation and Conservation", The Pennsylvania University Studies No. 23, University Park, 71 pp.

Peczenik, A., et al., 1956, "An Apparently Waterborne Outbreak of Infectious Hepatitis", Am J. Pub. Health, 46, 1009.

Preul, H. C. and Schroepfer, G. J., 1968, "Travel of Nitrogen in Soils", J. Water Pollution Control Federation, 40, 1, 30-48.

Robeck, G., 1966, "Ground Water Contamination Studies at the Sanitary Engineering Center", Ground Water Contamination, Proc. 1961 Symp., U. S. Dept. of Health, Education and Welfare, Technical Report W61-5.

Rudolfs, W., Faik, L. L., and Ragotzkie, R. A., 1950, "Literature Review on the Occurrence and Survival of Enteric, Pathogenic and Relative Organisms in Soil, Water, Sewage and Sludges, and on Vegetation" Sewage and Industrial Wastes, 22, 10, 1261-1281.

Santee Recreation Project, Santee, California, Final Report, 1967, U. S. Dept. of Interior, F. W. P. C. A., Cincinnati, Ohio, Publ. WP-20-7.

Sawyer, C. N., 1947, "Fertilization of Lakes by Agricultural and Carbon Drainage", J. New England Water Works Assn., 61, 2, 109-127.

Sawyer, C. N., 1952, "Some New Aspects of Phosphates in Relation to Lake Fertilization", Sewage and Industrial Wastes 24, 6, 768-776.

Stauffer, R. S., 1942, "Runoff Percolate and Leaching Losses from Some Illinois Soils", J. Amer Soc. Agon., 34, 830-835.

Stiles, C. W., Crohurst, H. R., and Thomson, G. E., 1927, "Experimental Bacterial and Chemical Pollution of Wells Via Ground Waters and the Factors Involved", U. S. Public Health Service Hyg. Lab. Bul. 147., 168 pp.

Stone, R., 1953, "Land Disposal of Sewage and Industrial Wastes", Sewage and Industrial Wastes, 25, 4, 406-418.

Sylvester, R. O., 1961, "Nutrient Content of Drainage Water For Forested, Urban and Agricultural Areas, Algae and Metropolitan Wastes, U. S. Public Health Service, San Engr. Center Publication TRW 61-3, 8--87.

U. S. Public Health Service Drinking Water Standards, 1962, U. S. Department of Health, Education, and Welfare, Public Health Service Publication No. 956.

U. S. Public Health Service, 1964, "Pollutional Effects of Stormwater and Overflows from Combined Sewer Systems", USPHS Publication No. 1246, 39 pp.

Van Donsel, D. J., Geldreich, E. E., and Clarke, N. A., 1967, "Seasonal Variations in Survival of Indicator Bacteria in Soil and Their Contribution to Stormwater Pollution", Appl. Microbiol., 15, 6, 1362-1370.

Wadleigh, C. H., 1968, "Wastes in Relation to Agriculture and Forestry", United States Department of Agriculture Miscellaneous Publication No. 1065, 112 pp.

Walton, G., 1951, "Survey of Literature Relating to Infant Methemoglobinemia Due to Nitrate - Contaminated Water, Am. J. of Public Health, 14, 8, 986-996.

Weber, A., 1959, "Chemical and Bacteriological Investigation of Ground Water During an Epidemic of Hepatitis, Osterr", Wasser, 10, 110 (1958); Public Health Engr. Abs., 39, 11, 27.

Weibel, S. R., Dixon, F. R., Weidner, R. B., and McCabe, L. J., 1964, "Waterborne Disease Outbreaks, 1946-1960", J. Am. Water Works Assoc., 56, 8, 947-958.

Weibel, S. R. , Weidner, R. B., Cohen, J. M. and Christianson, A. G., 1966, "Pesticides and Other Contaminants from Rainfall and Runoff" J. Am. Water Works Assn., 58, 8, 1075.

Welch, L. F. and Bomke, A. A., 1969, "Potential for Nitrogen Loss Through Tile Drains", Proc. 1969 Fertilizer Conferences, University of Illinois, Urbana, 37-39.

Iversen, Karsten, 1932, "Nitrogen Loss Under Application of Stable Manure", Tidsskr. Planteavl. 38, 1-74.

Volk, G. M., 1961, "Gaseous Loss of Ammonia from Surface-Applied Nitrogenous Fertilizers", Agriculture and Food Chemistry, 9 (4), 280-283.

5.2 Effect on Soils and Crops

Allen, W. A., 1938, "Conditioning and Sale of Sludge as Fertilizer", Sewage Works Journal, 10, 6, 945.

Baetz, C. O., 1942, "Sewage Treatment at Appleton", Water Works and Sewage, 89, 6, 273.

Capp, J. P. and Engle, C. F., 1967, "Fly Ash in Agriculture", Bur. of Mines Info. Circ., 8348, 210-220.

Crowther, E. M. and Bunting, A. H., 1942, "The Manurial Value of Sewage Sludges", J. Inst. Sew. Purif., 13-28.

Crowther, E. M. and Bunting, A. H., 1944, "The Manurial Value of Sewage Sludges, 2", J. Inst. Sew. Purif., 46-58.

Dean, Robert B., 1968, "Ultimate Disposal of Waste Water Concentrates to the Environment", Environmental Science and Technology, 2, 12, 1079-1086.

Heintze, S. G. and Mann, P. J., 1947, "Soluble Complexes of Manganic Manganese", Journal of Agricultural Science, Volume 37, pp 23-26.

Hodgson, D. R. and Holliday, R., 1966, "The Agronomic Properties of Pulverized Fuel Ash", Chemistry and Industry, 785-789.

Holliday, R., Hodgson, D. R., and Cope, F., 1958, "Plant Growth on Fly Ash", Nature, 181, 1079.

Hommon, C. C., 1931, "Imhoff Tank Sludge as Farm Fertilizer", Public Works, 62, 69-72.

Lagerwerff, J. V., 1967, "Heavy Metal Contamination of Soils", Soil and Water Conservation Res. Div., A. R. S., Beltsville, Maryland.

Lunt, H. A., 1959, "Digested Sewage Sludge for Soil Improvement", Connecticut Experiment Station Bulletin, 622, 1-30.

Lunt, H. A., 1953, "The Case for Sludge as a Soil Improver", Water and Sewage Works, 100, 8, 295-301.

Merz, R. C., 1959, "Utilization of Liquid Sludge", Water and Sewage Works, 11, 489-493.

Miller, J. R., 1966, "Soil-Plant Relationships to Consider for Sewage Sprinkler Irrigation Systems Used for Waste Disposal", Compost Science, 7, 2,22.

Miodusjewskie, W. and Hinesly, T, 1968, Unpublished Data.

Morton, J., 1966, "Discussion of 'Sludge Treatment and Disposal at Maple Lodge' by Kershaw, A. M. and R. Wood", Inst. of Sewage Purif., Jour. and Proc., Part I, 75.

Rees, W. J. and Sidrak, G. H., 1956, "Plant Nutrition on Fly-Ash", Plant and Soil VIII, 2, 141-159.

Rudolfs, W., 1928, "Sewage Sludge as Fertilizer", Soil Sci., 26, 455-458.

Salter, P. and Williams, J. B., 1967, "Effects of Pulverized Fuel Ash on the Moisture Characteristics of Soils", Nature, 213, 1157-1158.

Scott, R. H., 1962, "Disposal of High Organic Content Wastes on Land", Water Pollution Control Federation Journ., 34, 9, 932-950.

Stone, A. R., 1962, "Disposal of Sludges on Land", Inst. Sewage Purif., Journ. and Proc., 424-430.

Troome, S., Barbier, G. and Chabannes, J., 1951, "Chlorosis, Caused by Lack of Mn, of Crops Irrigated with Filtered Water from Paris Sewers", Biol. Abstracts, Vol 125, Part 4, Abstract Number 34985.

Van Kleeck, L., 1943, "Sludge as Fertilizer in the War Emergency", Water Works and Sewage, 90, 177-178.

5.3 Other Environmental Effects

Anon., 1949, "Control of Domestic Fly Breeding in Digested Sludge - Use of BHC Dust", Klipfuntein Organic Products, Public Health (Johannesburg), 13, 153.

Anon., 1968, "Land Reclamation Progress", The Seventh Wonder, Metropolitan Sanitary District of Greater Chicago, 4, 2, 3.

Cooke, W. B., 1954a, "Fungi in Polluted Water and Sewage, I", Sewage and Industrial Wastes, 26, 539-549.

Cooke, W. B., 1954b, "Fungi in Polluted Water and Sewage, II", Sewage and Industrial Wastes, 26, 661-674.

Cooke, W. B., 1954c, "Fungi in Polluted Water and Sewage, III", Sewage and Industrial Wastes, 26, 790-794.

Cram, E. B., 1943, "The Effect of Various Treatment Processes on the Survival of Helminth Ova and Protozoan Cysts in Sewage", Sewage Works Journal, 15, 6, 1119-1138.

Dixon, F. R. and McCabe, L. J., 1964, "Health Aspects of Wastewater Treatment", J. Water Pollution Control Federation, 36, 984.

Evans, J. O., 1968, "Ultimate Disposal and Soil Improvement". F. W. P. C. A.

Federation of Sewage Works Association Manual of Practice, No. 2, 1946, Utilization of Sewage Sludge as Fertilizer, Water Pollution Control Federation, Washington, D. C., 120 pp.

Glaser, J. R. and Ledbetter, J. O., 1967, "Size and Numbers of Aerosols Generated by Activated Sludge Aeration", Water and Sewage Works, 114, 6, 219-221.

Gotaas, H. B., 1956, "Composting", World Health Organization Monograph Series No. 31, Geneva, 205 pp.

Greenberg, B., Vatela, G., Bornstein, A., and Hernandez, H., 1963, "Salmonellae from Flies in a Mexican Slaughterhouse", American Journal of Hygiene, 77, 177-183.

Kershaw, M. A. and Wood, R., 1966, "Sludge Treatment and Disposal at Maple Lodge", J. and Proc. Inst. Sew. Purif., Part I, 75-93.

Ledbetter, J. O, 1964, "Air Pollution from Aerobic Waste Treatment", Water and Sewage Works, 111, 1, 62.

Lewin, V. H., 1968, "Sewage Sludge Disposal - Back to the Land", Effluent and Water Treatment Journal, 8, 1, 21-28.

[illegible], R. C., 1959, "Application of Liquid Sludge", Water and Sewage Works, 106, 11, 489-493.

Napolitano, P. J. and Row, D. R., 1966, "Microbial Content of Air Near Sewage Treatment Plants", Water and Sewage Works, 113, 12, 480-483.

Nusbaum, I. and Cook, L., Jr., 1960, "Making Topsoil with Wet Sludge", Wastes Engineering, 31, 438-440.

Pokrovsky, S. N. and Zina, G. G., 1938, "Flies as Carriers of Helminth Eggs under Natural Conditions", Meditsinskaya Parazitologiya i Parazitarnye Bolezni, 7, 26.

Quarterman, K. D., Mathis, W., and Kilpatrick, J. W., 1954, "Urban Fly Dispersal in the Area of Savannah, Georgia", Journal of Economic Entomology, 47, 3, 405.

Randall, C. W. and Ledbetter, J. O., 1966, "Bacterial Air Pollution from Activated Sludge Units", Paper presented at the Am. Ind. Hygiene Conf., Pittsburg, May, 1966.

Rawn, A. M. and Garber, W. F., 1950, "Sand Beds Versus Shallow Lagoons for Sludge Drying", Water and Sewage Works, 97, 287.

Reploh, H. and Handlose, M., 1957, "Investigations on the Spread of Bacteria Caused by Irrigation with Waste Water", Arch. Hyg. Berl., 141, 632.

Rudolfs, W., Falk, L. L., and Ragotzkie, R. A., 1950, Literature Review on the Occurrence and Survival of Enteric, Pathogenic, and Relative Organisms in Soil, Water, Sewage, and Sludges, and on Vegetation. I. Bacterial and Virus Diseases", Sewage and Industrial Wastes, 22, 10, 1261-1281.

Rudolfs, W., Falk, L. L, and Ragotzkie, R. A., 1951, "Contamination of Vegetables Grown in Polluted Soil", Sewage and Industrial Wastes, 23,
"I. Bacterial Contamination", 3, 253-268.
"II. Field and Laboratory Studies on Endamoeba cysts", 4, 478-485.
"III. Field Studies on Ascaris Eggs", 5, 656-660
"IV. Bacterial Decontamination", 6, 739-751.
"V. Helminthic Decontamination", 7, 853-860.
"VI. Application of Results", 8, 992-1000.

Schoof, H. F., and Siverly, R. E., 1954, "Urban Fly Dispersion Studies with Special Reference to Movement Pattern of Musca Domestica", American Journal of Tropical Medicine and Hygiene, 3, 539-547.

U. S. Public Health Service Report, 1964, "Activities Report, Basic and Applied Sciences Branch, Division of Water Supply and Pollution Control", U. S. Dept. Health, Education and Welfare, July 1, 1963 - June 30, 1964, as described by Hanks, 1967.

Webb, J. E., and Graham, H. M., 1956, "Observations on Some Filth Flies in the Vicinity of Fort Churchill, Manitoba, Canada, 1953-54", Journal of Economic Entomology, 49, 5, 595.

6.0

REFERENCES: ENVIRONMENTAL CONSIDERATIONS FOR REGIONAL OCEAN DISPOSAL SYSTEMS

Fair, G. M. and Geyer, J. C., 1954, Water Supply and Wastewater Disposal, John Wiley and Sons, New York, 973 pp.

Gunnerson, C. G., 1961, "Mineralization of Organic Matter is Santa Monica Bay, California, Symp. of Mar. Microbiology, Chicago, April, 1961.

Ketchum, B. A., Vaccaro, R. F. and Corwin, N., 1958, "The Annual Cycle of Phosphorus and Nitrogen in New England Coastal Waters", J. Mar. Res., 17, 282-301.

Ketchum, B. A. and Keen, D. J., 1955, "The Accumulation of River Water Over the Continental Shelf Between Cape Cod and Chesapeake Bay", Papers in Marine Biology and Oceonography, Suppl. to Vol. 3 of Deepsea Research, Pergamon Press, pp 346-357.

McKee, J. E. and Wolf, H. W., 1963, "Water Quality Criteria".

Metcalf, T. G. and Stiles, W. C., 1968, "Viral Pollution of Shellfish in Estuary Water", Proc. ASCE, SED SA4, 6063.

Moore, B., 1960, "The Risk of Infection Through Bathing in Sewage-Polluted Water", Proc. First Int. Conf. on Waste Disposal in the Mar. Invir., Pergamon Press, New York.

Pringle, B. H. and Shuster, C. N., Jr., 1967, "A Guide to Trace Metal Levels in Shellfish", Informal report, Northeast Marine Health Sciences Laboratory, Narragansett, Rhode Island.

Redfield, A. C., 1958, "The Biological Control of Chemical Factors in the Environment, Am. Scientist, 46, 205.

7.0

REFERENCES: TREATMENT OF WASTE MATERIALS

7.2 Treatment for Communicable Disease Agents

Allen, M. and Soike, K., 1966, "Sterilization by Electrohydraulic Treatment", *Science*, *154*, 155-157.

Allen, M. and Soike, K., 1967, "Disinfection by Electrohydraulic Treatment", *Science*, *156*, 524-525.

Anderson, M. S., 1959, "Fertilizing Characteristics of Sewage Sludge", *Sewage and Industrial Wastes*, *31*, 678.

Berg, G., Ed., 1965, *Transmission of Viruses by the Water Route*, Interscience Publishers.

Birkeland, J., 1949, *Microbiology and Man*, Appleton-Century-Crofts, Inc., New York.

Black, A. P., Kinman, R. N., Thomas, W. C., Jr., Freund, G., and Bird, E. D., 1965, "Use of Iodine for Disinfection", *Journal American Water Works Association*, *57*, 11, 1401-1421.

Chambers, C. W., 1961, "The Germicidal Efficiency of Silver, Iodine, and Quaternary Ammonia Compounds", *Proceedings Third Sanitary Engineering Conference*, University of Illinois, Urbana, 29-42.

Connell, C. H., Dreyer, D. A., and Berg, E. J. M., 1958, "Reduction of Coliform Bacteria in Sewage Sludge by Halogens", *Sewage and Industrial Wastes*, *30*, 5, 634-645.

Davis, B. B., 1948, "Principles of Sterilization", *Bacterial and Myotic Infections of Man*, ed. by R. J. Dubos, J. B. Lippincott Co., Philadelphia.

Eighth Research Report of the Metropolitan Water Board, London, England.

Faber, H. A., 1961, "Water Disinfection", International Water Supply Congress, International Water Supply Association, London.

Furczyk, A. A., 1955, "High Speed Sludge Heating", *Public Works Mag.*, July.

Gilliland, W. E. and Speck, M. L., 1967, "Mechanism of the Bactericidal Action Produced by Electrohydraulic Shock", *Applied Microbiology*, 15, 5, 1038-1044.

Hoover, Charles P., 1924, Journal American Water Works Association, Vol. 11, May.

Hoover, Charles P., 1962, Water Supply and Treatment, Ninth Ed., National Lime Association, Washington, D. C.

Kaplan, S. and Melnick, J. L., 1954, "Effect of Milk and Other Dairy Products on the Thermal Inactivation of Coxackie Viruses", Am. J. of Public Health, 44, 9, pp 1174-1184.

McKinney, R. E., Longley, H. E., and Tomlinson, H. D., 1958, "Survival of Salmonella Typhosa during Anaerobic Digestion", Sewage and Industrial Wastes, 30, 12, 1469-1477.

Morris, J. C., 1966, "Future of Chlorination", Journal American Water Works Association, 58, 11, 1475.

Public Health Service Drinking Water Standards, 1962, Public Health Service Publication No. 956, 61 p.

Riehl, M. L, Weiser, H. H., and Rheins, B. B., 1952, "Effect of Lime-Treated Water Upon Survival of Bacteria", Jour. Am. Water Works Assoc., 44, 5, pp 466-470, May.

Stumm, W., 1958, "Ozone as a Disinfectant for Water and Sewage", Journal Boston Society of Civil Engineers, 45, 1, 68.

Tyler, R. G., Orlob, G. T., and Williams, F. W., 1950, "Chlorination of Raw and Digested Sludge", Sewage and Industrial Wastes, 22, 7, 875-883.

Vater, W., 1969, "European Practices in Sludge Dewatering and Disposal", Paper presented at special lecture series on "Advances in Water Quality Improvement - Physical and Chemical Processes", University of Texas at Austin, April.

Water Pollution Control Federation Manual of Practice No. 4, 1951, "Chlorination of Sewage and Industrial Wastes", 84 pg.

Wattie, E., and Chambers, C. W., 1943, "Relative Resistance of Coliform Organisms and Certain Enteric Pathogens to Excess-Lime Treatment", Jour. Am. Water Works Assoc., 35, 6, pp 709-720, June.

Wyss, O. and Stockton, F. D., 1947, "The Germicidal Action of Bromine", Archives of Chemistry, 12, 267.

7.3 Treatment for Nitrogen Removal

Eliassen, R. and Tchobanoglous, G., 1968, "Removal of Nitrogen and Phosphorus", PaperPresented at 23rd Purdue Industrial Waste Conference, Lafayette, Indiana.

Farrell, J. B., Stern, G. and Dean, R. B., 1968, "Nitrogen Removal from Wastewaters", Unpublished Paper.

Federal Water Pollution Control Administration, 1968, "Summary Report Advanced Waste Treatment, July, 1964 - July, 1967", F. W. P. C. A. Publication WP-20-AWTR-19, 96 pg.

Johnson, W. K. and Schroepfer, J. G., 1964, "Nitrogen Removal by Nitrification and Denitrification", Journal W. P. C. F., Vol 36:8, pp 1015-1036.

Mercer, B. W., Ames, L. L. and Touhill, C. J., 1968, "Ammonia Removal from Wastewater by Selective Ion Exchange", Paper Presented at Meeting of the American Chemical Society, San Francisco.

Slechta, A. F. and Culp, G. L., 1967, "Water Reclamation Studies at the South Tahoe Public Utility District", Journal Water Pollution Control Federation, 39, 5, 787-814.

8.0

REFERENCES: LAND DISTRIBUTION SYSTEMS

Dodson, R. E. and Stone, R., 1962, "Advances in Sludge Disposal", Journ. of Sanitary Engineering Division ASCE, 88 (SA4), 71-72.

Law, J. P., Jr., 1968, "Agricultural Uilization of Sewage Effluent and Sludge", An Annotated Bibliography, Fed. Water Pollution Control Adm., U.S. Dept. of the Interior, CWR-2, 87 pages.

Scanlon, A. J., 1957, "Utilization of Sewage Sludge for the Production of Topsoil", Sewage and Industrial Wastes, 29, (8), 944-950.

Scott, R. H., 1962, "Disposal of High Organic Content Wastes on Land", Journ. Water Pollution Control Federation, 34, (9), 932-950.

Wolfel, R. M., 1967, "Liquid Digested Sludge to Land Surfaces, Experiences at St. Mary's and Other Municipalities in Pennsylvania", Paper Presented at 39th Annual Conf. of the Water Pollution Control Assoc. of Penna., August.

9.0

REFERENCES: OCEAN DISTRIBUTION SYSTEMS

Abraham, G. , 1963, "Jet Diffusion in Stagnant Ambient Fluid", Delft Hydraulics Laboratory, Pub. No. 29, Delft, The Netherlands.

Haight, F. J. , 1942, "Coastal Currents Along the Atlantic Coast of the United States, Coast and Geodetic Survey, Sp. Pub. 230.

Lanskaya, L. A. , 1961, "The Division Rates of Plankton Algae of the Black Sea in Cultures", Marine Microbiology , p. 127.

Okubo, Akira, 1968, "Summary of Our Present Knowledge of the Physical Processes of Mixing in the Ocean and Coastal Waters, and a Set of Practical Guidelines for the Application of Existing Diffusion Equations in the Preparation of Nuclear Safety Evaluations of the Use of Nuclear Power Sources in the Sea", Topical Report AEC, Contract AT (30-1)-3109, Chesapeake Bay Institute, The Johns Hopkins University, Ref. 69-1.

Woods Hole Oceanographic Institution, 1961, "Biological, Chemical and Radiochemical Studies of Marine Plankton", Appendix C to Ref. No. 61-6, AEC Contract AT(30-1)-1918.